FEASIBILITY OF MITIGATING ATMOSPHERIC ACIDIFICATION OF A SMALL STREAM IN WESTERN MARYLAND

CHESAPEAKE BAY AND WATERSHED PROGRAMS
MONITORING AND NON-TIDAL ASSESSMENT
CBWP-MANTA- EA-02-1

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Feasibility of Mitigating Atmospheric Acidification of a Small Stream in Western Maryland

FINAL REPORT

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Foreward

This report describes research and monitoring activities that were conducted under the auspices of Maryland Department of Natural Resources (MDNR). Funding for the project was provided by Resource Assessment Service, Monitoring and Non-Tidal Assessment Division, MDNR, under the direction of Dr. Ronald J. Klauda, through grant no. 07-4-30616 to Appalachian Laboratory, University of Maryland Center for Environmental Science, and the principal investigator, Dr. Keith N. Eshleman. The authors thank Mr. Matt Kline for collecting benthic macroinvertebrate samples at the study sites and acknowledge the constructive reviews of the first draft of the report by Mr. Paul Kayzak and an anonymous referee. This report is technical contribution no. TS-354-02 from University of Maryland Center for Environmental Science.

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Executive Summary

Several recent studies have concluded that the reductions in atmospheric sulfur deposition loading mandated under the 1990 Clean Air Act Amendments will be insufficient to restore the alkalinity of a large percentage of streams and lakes in the eastern United States in the near future. Therefore, efforts to locally mitigate the effects of acid deposition on surface water receptors will need to continue for at least a few decades in order to prevent or ameliorate acidification of some aquatic resources. The goal of this project was to provide a recommendation to Maryland Department of Natural Resources (MDNR) regarding the feasibility of using alkaline groundwater pumping (or an alternative technique) to mitigate acidification of a small stream in western Maryland. The Murley Run watershed, a 1,127 ha subwatershed of the Herrington Creek drainage basin in Garrett County, Maryland, had been previously identified by MDNR as an excellent candidate for water quality restoration using a local acid mitigation scheme.

The specific objectives of the project were to: (1) review the current scientific literature on alternative acid deposition mitigation strategies for lotic systems; (2) conduct baseline water chemistry monitoring within the Herrington Creek watershed that was necessary to assess the feasibility of mitigation; (3) perform a baseline qualitative and quantitative benthic macroinvertebrate survey within the Murley/Bull Glade Run watershed; (4) determine the feasibility of acid deposition mitigation, particularly alkaline groundwater pumping, within the Murley/Bull Glade Run watershed; and (5) design a restoration plan, including short-term testing, maintenance, and monitoring. Objective (4) included several sub-objectives: (4a) complete a review of the subsurface hydrogeological environment of the Murley and Bull Glade Run watersheds using existing data; (4b) characterize the well production rates and groundwater

quality for a group of existing wells in the vicinity of the watershed; and (4c) assess logistical constraints on the use of alkaline groundwater pumping as a mitigation strategy.

The results of the multi-year water chemistry study indicated that Murley Run and its tributaries have been *chronically* acidified as a result of a long history of atmospheric sulfur deposition. Streamwater ANC values based on very accurate Gran titrations were found to be negative throughout most of the year, with pH (closed system) values typically less than 5.0 in the headwaters of the basin; under all but the lowest flow conditions, both total and exchangeable reactive aluminum concentrations typically exceeded 500 μg/L in these upper reaches. Dissolved organic carbon concentrations in the streams were typically less than 2 mg/L, consistent with the interpretation that natural organic acidity is not a dominant contributor to the strong acidity of the Murley streams. Calcium and magnesium concentrations were also observed to be quite low—consistent with an explanation that the surface waters in the Murley Run watershed are geochemically-sensitive to atmospheric inputs of mineral acids (particularly sulfuric acid). Data from two storm events revealed that the Murley streams do not undergo severe *episodic* acidification, however. This transient process is more commonly observed in stream systems that have ANC values greater than 25 μeq/L.

An investigation of the hydrogeological system underlying the watershed revealed that one or more relatively deep wells in the area would likely be able to supply groundwater in sufficient quantity and of alkaline quality to support the alkaline groundwater injection into the stream. Since shallow wells in the area were deemed to produce water that is substantially less alkaline and too heavily laden with iron to be used in such a mitigation effort, groundwater wells would need to be cased in a way that would prevent the shallower groundwater from entering the wells. A simple mixing model was used to evaluate the efficacy of several different injection schemes (both constant and variable rate). The model demonstrated that a small storage reservoir

(4,000-8,000 gallons—about the size of a small backyard swimming pool)—could be used to maintain relatively high streamwater ANC values during high flow conditions that typically accompany winter and spring rain and snowmelt events.

We concluded that it would be feasible to mitigate acidification of the Murley/Bull Glade system using an alkaline groundwater pumping scheme and that both the installation and small "footprint" of such a system (limited to a small storage reservoir and well house/instrument shelter) within the state forest would cause minimal intrusion and disturbance of the forest land. The primary limitation of the system is the relatively high up-front cost associated with bringing line AC power to the site; estimated total project costs (construction, operation, maintenance, and monitoring) over a 10-year period were found to be approximately $265,000. Other distributed methods for producing electrical power to operate the groundwater pumping and injection system (e.g., fuel cells) will likely be available within a few years and these options would expectedly reduce the capital costs associated with extending line power to the site. Regardless, a project of this type might still be financially feasible if it could be integrated into a larger research, monitoring, and demonstration project focused on local mitigation of surface water acidification and restoration of a freshwater ecosystem.

Introduction

Western Maryland is located in a region of the United States that has received chronic atmospheric deposition for much of the last century with current acidic deposition rates being among the highest in North America (Lynch *et al.*, 1996). It is now known that acidic deposition has the potential to acidify surface waters in regions where systems are pre-disposed due to the presence of geological substrates that provide little acid neutralizing capacity (Schindler, 1988). It has been further established that acidification can take either of two forms: (1) *chronic* acidification, in which surface water acid neutralizing capacity (ANC) is critically low (usually negative) for extended periods of the year; and (2) *episodic* acidification, in which surface water ANC is transiently depressed to low levels only during periods associated with stormflow runoff from rainfall and snowmelt processes (Wigington *et al.*, 1990). Many studies performed in the 1970's and 1980's suggested that the pH and ANC of surface waters in Europe and North America had declined over a period of about 50 years as a result of chronic acidification (Beamish and Harvey, 1972; Beamish *et al*., 1975; Wright and Gjessing, 1976; Kaufmann *et al.* 1988, Sullivan *et al.*, 1988, Baker *et al.*, 1990, Bricker and Rice, 1993), causing the loss of native fish populations and other biological effects.

The Title IV Clean Air Act Amendments of 1990 (CAAA-90) mandated substantial reductions in both sulfur dioxide and nitrogen oxide emissions from power plants by the year 2000. The primary goal for sulfur dioxide emissions was a 10 million ton reduction below 1980 emission levels by the year 2000. The primary goal for nitrogen oxide emissions was a 2 million ton reduction from 1980 emission levels by the year 2000. There have already been substantial reductions in sulfur dioxide emissions through the Phase I regulations (Public Law 101-549). In fact, Lynch *et al.* (1996) analyzed the U.S. national precipitation monitoring data and concluded that these reductions in sulfur dioxide emissions have led to lower sulfate concentrations in precipitation in the eastern United States, especially within the Ohio River Valley and Mid-Atlantic region. They further concluded that resultant emission reductions from the 1990 Clean Air Act Amendments have reduced acidic deposition in the eastern United States.

Since reductions in emissions of sulfur dioxide and nitrogen oxide can be expected to yield lower acidic deposition rates in the United States, questions now remain as to whether streams that have become acidified will be able to recover and how long this recovery might take. Modeling results from the Nitrogen Bounding Study (NBS) projects that sulfur deposition reductions mandated by CAAA-90 will benefit sensitive surface waters in the mid-Appalachians by the year 2040 (U.S.E.P.A., 1997). Additionally, Bulger *et al.* (1998) used mathematical models to predict whether Virginia headwater brook trout streams would continue to support brook trout by the years 2011 and 2041 using a variety of emission reduction scenarios. Bulger *et al.* (1998) concluded that a 70 percent reduction in emissions would be required to retain 50% of Virginia trout streams that are currently not acidic (ANC > 50 μeq/L). Therefore, streams that have become chronically acidic (ANC < 0 μeq/L) as a result of acid deposition cannot be expected to recover in the near future even after these recently mandated reductions in acid loadings. The Maryland Critical Loads Study (Janicki *et al.*, 1991) also concluded that the critical loads within several watersheds in Maryland will continue to be exceeded even after these federally-mandated emissions reductions have been fully implemented.

Consequently, efforts to mitigate acidic deposition on receptors, in addition to sources, will need to continue for at least the next few decades in order to prevent or ameliorate acidification of streams. Ideally, mitigation efforts on receptors should be cost-effective, reliable, unobtrusive, low-maintenance, and should cause few or no undesirable side effects. The Murley Run watershed, a 1,127 hectare subwatershed in the Herrington Creek drainage basin in Garrett County, Maryland (Figure 1), has been identified as an excellent candidate for water quality restoration using a local acid mitigation scheme for a variety of reasons. The watershed is located in an area of the state that has historically been sensitive to the effects of acid deposition and this acid-sensitivity will likely continue even after full implementation of CAAA-90 since the natural watershed buffering capacity is now completely depleted. Since most of the watershed is located within state forest boundaries, future land use changes within the watershed are controlled and a restored stream could provide opportunities for extended public use of the area. Further, ecological assessment has shown that Murley Run—though currently devoid of fish—has a diverse benthic macroinvertebrate population that would be necessary to support a native brook trout fishery. The stream's excellent physical habitat features appear to meet the requirements of many fish species including brook trout (Southerland and Volstad, 1996). Although not a direct objective of this project, a potential secondary benefit of water quality restoration performed in the Murley Run watershed would include significant improvements in Herrington Creek below its confluence with Murley Run. Although water temperatures in Herrington Creek are too high in the summer months to support trout (Southerland and Volstad, 1996), it could be managed as a warmwater fishery or as a spring "put-and-take" area for trout.

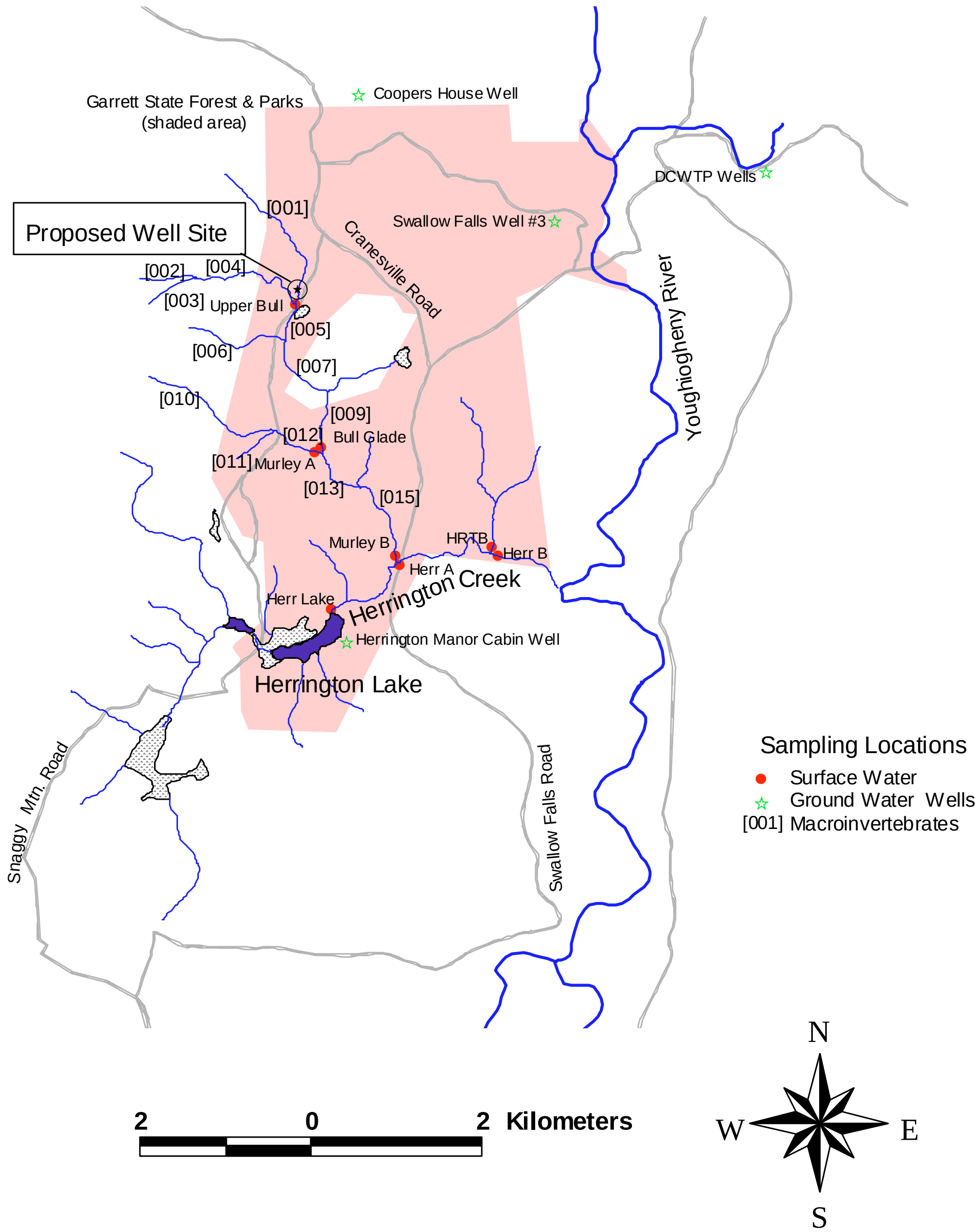

Figure 1. The Herrington Creek, Murley Run, and Bull Glade Run study area in Garrett County, western Maryland. Locations of proposed well site, existing wells, and water sampling stations are also shown.

Objectives

The overall goal of this phase of the project was to provide a recommendation to Maryland Department of Natural Resources (MDNR) on the feasibility of utilizing alkaline groundwater pumping (relative to several alternative techniques) to mitigate the atmospheric acidification of a small stream in western Maryland. The specific objectives of the project were to:

(1) review the current scientific literature on alternative acid deposition mitigation strategies for lotic systems;

(2) conduct baseline water chemistry monitoring within the Herrington Creek watershed that was necessary to assess the feasibility of mitigation;

(3) perform a baseline qualitative and quantitative benthic macroinvertebrate survey within the Murley/Bull Glade Run watershed;

(4) determine the feasibility of acid deposition mitigation, particularly alkaline groundwater pumping, within the Murley/Bull Glade Run watershed; and

(5) design a restoration plan, including short-term testing, maintenance, and monitoring.

Objective (4) included several sub-objectives:

(4a) complete a review of the subsurface hydrogeological environment of the Murley and Bull Glade Run watersheds using existing data;

(4b) characterize the well production rates and groundwater quality for a group of existing wells in the vicinity of the watershed; and

(4c) assess logistical constraints on the use of alkaline groundwater pumping as a mitigation strategy.

Review of Acid Deposition Receptor Mitigation Strategies

A variety of methods have been developed to mitigate the effects of acid deposition in streams and restore suitable water quality for aquatic biota. Most techniques have employed the introduction of lime or other alkaline materials into the stream, including mechanical dosers, diversion wells, limestone barriers, and rotary drums (Olem, 1991). Mechanical dosers have been used in western Maryland to mitigate the effects of acid mine drainage (AMD), releasing either powdered or slurried lime into streams. The utility of mechanical dosers for acid mitigation is limited for many reasons, including frequent mechanical failures, high routine maintenance needs (i.e., refilling of limestone on a regular basis), and continuous high operating costs. Additionally, mechanical dosers often cause some undesirable water quality problems, such as decreases in transparency, increases in sediment load, and increases in water temperature. The increased sediment load can also cause problems by filling of interstitial voids in the benthic substrate, affecting the benthic macroinvertebrate community and interfering with fish reproduction.

Diversion wells have occasionally been used to mitigate acidic stream conditions. A diversion well is a container filled with alkaline material that is located along the stream bank or in the bottom of the streambed. Water is diverted from the stream and directed to the bottom of the well, which then moves upward through the alkaline material in the well. The diverted stream water is then discharged over the lip of the well or through an outlet pipe to the adjacent stream (Olem, 1991). Diversion wells are of limited utility because they must be refilled with limestone on a weekly basis, resulting in continuous maintenance and high operating costs.

Constructed wetlands have also been proposed as a way to improve stream water quality in Murley Run (Olem and Jacobsen, 1994). They are designed to simulate the treatment of water as it passes through natural wetlands, although few studies have documented generation of ANC from such systems.

Watershed liming is another technique that has been proposed to mitigate the effects of acid deposition in western Maryland streams. A pilot study was performed in 1990 within the Alexander Run watershed which proved watershed liming to be unsuccessful (Price *et al.*, 1993). Since the bedrock geology and water chemistry within the Alexander Run and Murley Run watersheds are very similar, we would not expect watershed liming to be effective at the latter site either.

In-stream addition of alkaline sand (ILS), a more recently developed technique, involves addition of calcareous sand directly into the stream channel using dump trucks. Streamflow transports the fine neutralizing material downstream, allowing it to dissolve in the water column, react with the acidic water, and gradually settle into natural sediment traps, creating a reservoir of alkaline material that is then available for neutralization of acidic water in the future. This technique, which has been successful in Pennsylvania, West Virginia and Virginia (Downey *et al.*, 1994; Zurbuch *et al.*, 1997; Sampsell, 1999), is attractive because it is relatively inexpensive and requires only periodic maintenance—usually only when the next dose needs to be applied (typically annually). Streams treated using the ILS technique have experienced increases in pH and ANC, as well as decreases in aluminum concentration. Stream profile, existing water chemistry, and flow regime each affects the degree of treatment using ILS (Downey *et al.*, 1994). Potential damage to benthic habitat at the application site and potential loss of buffering capacity during extremely high flow events resulting in decreases in pH and ANC below levels suitable for biota should both be considered before employing such a technique.

Alkaline groundwater pumping has also been used to mitigate the effects of acid deposition. In this technique, groundwater with sufficient alkalinity to neutralize acidity and thus improve water quality is pumped at a specified rate into a stream or lake. This technique has been used successfully in the Linn Run watershed located on the Appalachian Plateau in southwestern Pennsylvania (Gagen *et al.*, 1989). In this restoration, three wells were drilled to depths of about 110-150 meters below the surface, penetrating the Pocono geologic formation. Groundwater pumping and discharge were shown to increase mean stream pH from 4.9 upstream of the wells to 6.0 in the treatment section. Mean dissolved aluminum also decreased dramatically below the treatment wells. This method may be particularly suitable for streams in western Maryland, since the technique has been proven successful in a region that is geologically similar. Alkaline groundwater pumping should be relatively maintenance-free, reliable, relatively cost-effective, unobtrusive, free of undesirable side effects, and easily regulated and adapted to local streamflow and water quality conditions.

Finally, conventional drinking water treatment based on dissolution of solid sodium hydroxide is another possible mitigation technique, but this option would likely require an investment in small-scale water treatment/chemical handling equipment and chemical expendables that is probably not economically feasible. As with alkaline groundwater pumping, this technique would likely be free of undesirable side effects and could be closely tailored to local streamflow and water quality conditions; maintenance and reliability are major unknowns, however.

Methods

Stream water quality was monitored on a monthly basis at eight sites within the Herrington Creek watershed (Figure 1):

(1) Upper Bull Glade Run (Upper Bull);

(2) Bull Glade Run just above its confluence with Murley Run (Bull);

(3) Murley Run above the confluence with Bull Glade Run (Murley A);

(4) Murley Run below the confluence with Murley Run (Murley B);

(5) Herrington Lake outlet (Herr Lake);

(6) Herrington Creek below it confluence with Murley Run (Herr A);

(7) Unnamed tributary to Herrington Creek (HRTB); and

(8) Herrington Creek just below its confluence with HRTB (Herr B).

Grab samples were collected at each station in 1-L cubitainers (that had been rinsed several times and filled in the laboratory with deionized water) and two 50-mL syringes following standard field protocols for acidification studies (U.S. E.P.A., 1989). All sample containers were rinsed at least three times with streamwater prior to filling and all samples were kept on ice in a cooler during transport to AL. Each cubitainer sample was filtered within 24 hours of sample collection into aliquots for determination of the following constituents: (1) major ions (i.e., sodium, potassium, magnesium, calcium, chloride, nitrate, and sulfate), (2) dissolved organic carbon (DOC), and (3) metals (iron and manganese). The DOC aliquots were preserved with 125 µL of phosphoric acid and the metals aliquots were preserved with 60 µL of Ultrex-grade nitric acid. Samples were collected at five sites from January 1997 through February 1998 and at all eight sites from January 1998 through June 2000. In addition, water chemistry data from one site (HRTB) were collected continuously throughout the study as part of another MDNR-sponsored project (Eshleman *et al.*, 2000).

All streamwater samples were analyzed according to standard laboratory methods appropriate for monitoring surface water quality in acid deposition studies (U.S.E.P.A., 1987) as follows:

(1) Grab sample pH was measured using the "closed system" method on syringe samples to minimize changes due to equilibration with atmospheric carbon dioxide.

(2) A Brinkmann-Metrohm auto-titration system was used for measurement of ANC. A standard volume of sample was titrated with dilute HCl following the measurement of ambient pH (open system) with electrometric pH detection following the acidimetric Gran titration technique. Only those titration points (typically 6-8 points) taken in the range of 4.7 to 3.5 were used to compute the Gran function and ANC.

(3) Specific conductance was measured using a conductivity cell and Yellow Springs (YSI) conductivity meter with manual temperature correction to 25°C.

(4) Dissolved organic carbon (DOC) was measured by infrared detection following UV-assisted persulfate oxidation on a Tekmar-Dohrmann Phoenix 8000 analyzer.

(5) Total and exchangeable reactive aluminum concentrations were measured using an automated fractionation/pyrocatechol violet (PCV) flow injection analysis technique on an Alpkem Flow Solution IV FIA system. Exchangeable reactive aluminum was determined by subtraction of the non-exchangeable fraction from the total concentration of PCV-reactive aluminum.

(6) Major anions were determined using a Dionex DX-500 ion chromatography system, equipped with electronic conductivity suppressor, autosampler, and a computer-based data acquisition/controller system. Anions were eluted using the AS-9HC analytical column.

(7) Sodium and potassium were measured by flame atomic emission spectroscopy on a Perkin-Elmer AAnalyst 800 Spectrometer. Magnesium and calcium were measured by flame atomic absorption spectroscopy on a Perkin-Elmer AAnalyst 800 Spectrometer. A flow injection analysis accessory was used to perform in-line addition of lanthanum chloride for ionization suppression.

(8) Iron and manganese were measured by graphite furnace atomic absorption spectroscopy on a Perkin-Elmer AAnalyst 800 Spectrometer.

Three of the sites were also sampled during a period of high stream discharge to quantify the temporal changes in stream water chemistry during stormflow conditions: (1) Bull Glade Run just above its confluence with Murley Run (Bull), (2) Murley Run above the confluence with Bull Glade Run (Murley A), and (3) Herrington Creek below the confluence with Murley Run (Herr A). Samples were collected using automatic Sigma samplers programmed to collect samples every 4 hours beginning at the time of hydrograph rise and ending when conditions return to pre-storm levels. Episode samples were analyzed for open pH, specific conductance, ANC, and reactive Al (total, exchangeable, and non-exchangeable fractions) using the methods described above.

Monthly stream stage and discharge measurements were made using a Marsh McBirney flow meter at the Murley A, Bull, and Herr A sites from July 1999 through June 2000; staff gages had been installed at each of the three sites at the beginning of the project. The instantaneous discharge measurements were related using linear regression to the continuous record of hourly discharge from the HRTB site in order to estimate a continuous record of flow for both Murley A and Bull just above their confluence.

In order to obtain baseline (i.e., pre-mitigation) qualitative and quantitative estimates of the benthic macroinvertebrate communities within the Murley Run watershed, three sets samples were collected at each reach along Murley Run from the confluence with Herrington Creek to reaches that cross Snaggy Mountain Road in the upper reaches of the watershed (13 sampling locations) in the spring of 2000 (Figure 1). D-frame aquatic net and course particulate organic matter (CPOM) samples were collected and processed from all stations on Murley Run using guidelines for rapid bioassessment protocols for use in streams and rivers as established by the U.S.E.P.A. (1989). Nine D-frame aquatic net samples using a 600 micron mesh net were combined and approximated a total sample area of 1 m^2. The CPOM samples consisted of several handfuls of litter. Because D-frame and CPOM samples only provide qualitative assessments of benthic populations, four 4-inch (82 cm) “T” samples (Mackie and Bailey, 1981) were also collected from each station to provide quantitative population estimates. All samples were collected from riffle zones at each station, primarily because it is typically the most productive habitat in stream ecosystems and normally supports the most diverse community. In addition, a number of pollution-sensitive taxa occur in the riffle/run habitat, especially taxa associated with scraper and filtering collector functional feeding groups.

Benthic samples were transferred to polyethylene bottles and preserved in 70% ethyl alcohol in the field. In the laboratory, as much debris as possible was removed from each sample. Each sample was then placed in a 5 cm x 5 cm gridded pan. The first 300 organisms were picked for identification from large D-frame and CPOM samples. Small samples were picked completely. All macroinvertebrates in the "T" samples were also picked completely. The "T" and D-frame samples were then sorted and identified to the lowest efficient taxon. The macroinvertebrates in the CPOM samples were classified into appropriate shredder, collector, scraper, and predator functional feeding groups.

Statistical analyses of "T" samples followed techniques from Ludwig and Reynolds (1988) and Plafkin *et al.* (1989). Hill's family of diversity measures (N, N1, and N2) are in units of number of species, and measure the effective number of species present in a sample. N is the number of all species in the sample regardless of their abundance, $N1 = e^{H'}$ (an expression of Shannon's Index) measures the number of abundant species, and $N2 = 1/\lambda$ (an expression of Simpson's Index) is the number of very abundant species. The effective number of species is a measure of the number of species in the sample where each species is weighted by its abundance. These indices (N1 and N2) are most useful in working with benthic macroinvertebrates since rarer species are not given as much weight. The measure of evenness, $E_5 = (1/\lambda) -1/(e^{H'} -1) = N2-1/N1-1$, is known as the modified Hill's ratio. E_5 approaches zero as a single species becomes more and more dominant in a community, which is affected by sample size. Richness Index 1, Margalef's Index, is expressed as $R1 = s'-1/\log_e N'$, where s' is the total number of aquatic taxa recognized and N' is the total number of identified aquatic organisms. The proportional numerical dominance of the most abundant taxon is expressed as follows: N_i /N' where N_i is the most abundant taxon. Means and standard errors of these data for each set of “T” samples were calculated.

The CPOM and D-frame aquatic net samples were analyzed using the RBP III methodology (Plafkin *et al.*, 1989; Klemm *et al.*, 1990; Stribling *et al.*, 1998). The ratio of shredder functional feeding group and total number of individuals collected was determined from the CPOM samples. Shredders are good indicators of toxicants that are adsorbed on leaf litter in the riparian zone and affect the microbial communities that colonize leaf surfaces or affect shredders directly.

Analysis of the D-frame aquatic samples consisted of calculating nine metrics as described by Stribling *et al.* (1998). The first metric, taxa richness (or species richness), is an index of stream health, since the number of genera or species usually rises with increasing water quality, habitat diversity, and habitat suitability. Some care must be employed using this metric in pristine first- or second-order, headwater streams, since diversity may be low normally, however.

A second metric, the EPT index, is calculated as the total number of distinct taxa within the orders Ephemeroptera, Plecoptera, and Trichoptera (Plafkin *et al*, 1989). The EPT index normally increases as water quality increases and is an important taxa richness index for three orders of aquatic insects (Ephemeroptera, Plecoptera, and Trichoptera) considered to be pollution sensitive. A third metric, Ephemeroptera taxa, reflects the ability of a stream to support this generally intolerant insect order. Organic enrichment and fine sediment on gravel substrates will reduce the diversity of the mayfly population. Hall *et al.* (1980) reported a reduction in species diversity and density in field and laboratory studies where a healthy stream was subjected to a reduced pH. Ephemeropteran and dipteran taxa proved most sensitive.

The fourth metric, Dipteran taxa, are variable in their tolerance to stress. High diversity is generally an indication of good water and habitat quality.

The fifth metric, percent Ephemeroptera, reflects the degree to which mayflies dominate the community and can indicate the extent to which this pollution-sensitive order can maintain a reproductive population. The presence of stresses will reduce the abundance of most mayfly groups.

The sixth metric, percent Tanytarsini of Chironomidae, is an indication of the degree to which this intolerant group of midges represents the total midge assemblage. A high percentage of Tanytarsini may indicate lower levels of anthropogenic stress. *Tanytarsus* dissimilis has shown evidence of inability to complete its life cycle when the pH was below 5.5.

The seventh metric, intolerant taxa, are generally specialists in habitat and water quality and the first to be eliminated by perturbations. Tolerance values were assigned to each taxon based on the work established by Hillsenhoff (1982, 1987). Values ranged from 0 to 10, with highly intolerant individuals rated at 0 and very tolerant rated at 10. Intolerant taxa for this metric have tolerance ratings from 0 to 3.

The eighth metric, percent tolerant taxa, is the percentage of individuals with tolerance ratings of 7 to 10. This metric tends to increase as perturbation increases. Intolerant taxa decrease as more tolerant, opportunistic taxa increase.

The ninth metric, percent collector gatherers, is the percentage of the macroinvertebrates that feed on detrital deposits or loose surface films. This functional feeding group can be expected to decrease with increasing stress. The nine metrics were assigned a score developed by Stribling *et al.* (1998). These scores were then combined and averaged to determine an index of biotic integrity (IBI) for the corresponding sampling station.

Results and Discussion

Surface Water Hydrology

Stage/discharge measurements for the three intensive sites are presented in Table 1. With the exception of August and September 1999 (an oversight by the field technician), monthly stage and discharge measurements were completed at the time of each monthly water sampling according to the project plan. While virtually all of the measured discharges were very low due to the extended drought in western Maryland, it was possible to predict the discrete monthly discharge values for both Murley A and Bull from the corresponding instantaneous discharge at the HRTB site modified by the ratio of their watershed areas (Figure 2). The statistical relationships between the predicted and observed values from the linear regressions have x-coefficients that are very close to unity, although the y-intercepts are greater than zero. These results suggest that the Murley A and Bull sites generate comparable amounts of runoff as the HRTB site under moderately high flow conditions, but their lowest baseflows are somewhat higher. Discharge at HRTB (combined with the watershed area ratio) was not found to be a satisfactory predictor of discharge at the Herr A site, however (Figure 3). It should be noted that the stream water quality and quantity of flow at Herr A are influenced by the flow controls at Herrington Lake, located approximately 1500 meters upstream. According to Garrett State Forest personnel, the level of water in the lake is lowered in the colder months (October – March) to manage the growth of lake vegetation, which left unchecked year round could have adverse effects on the fish population in the lake. During these colder months, water is discharged through the stand pipe beneath the dam. Magnitude of flow through this pipe depends upon the level of water in the lake. During the warmer months (April–September), the pipe beneath the dam is closed and the lake level rises and water discharges via the spillway.

Table 1. Staff Levels and Discharge Estimates for Dates of Surface Water Sampling

Date	Bull Staff (cm)	Bull Discharge (m3/sec)	Murley A Staff (cm)	Murley A Discharge (m3/sec)	Herr A Staff (cm)	Herr A Discharge (m3/sec)	HRTB Staff (cm)	HRTB Discharge (m3/sec)
7/8/99	9.5	0.00238	4.0	0.00012	10.0	0.00838	12.0	0.00000
8/10/99	12.0	N/A[1]	6.5	N/A[1]	12.0	N/A[1]	14.0	0.00000
9/14/99	7.5	N/A[1]	5.0	N/A[1]	9.0	N/A[1]	12.0	0.00000
10/13/99	15.0	0.00830	11.0	0.00268	15.0	0.03233	22.0	0.00160
11/18/99	20.0	0.02129	17.0	0.01297	18.0	0.12881	21.0	0.00110
12/16/99	32.0	0.25746	29.0	0.12872	44.0	N/A[1]	32.0	0.09187
1/18/00	25.0	0.10297	23.0	0.04088	28.0	0.45155	27.0	0.02044
2/22/00	35.0	0.41274	33.0	0.15413	49.0	1.63482	27.0	0.02044
3/15/00	28.0	0.14702	28.0	0.07632	26.0	0.42356	23.0	0.00374
4/25/00	29.0	0.17838	26.0	0.10406	37.0	1.08638	23.0	0.00374
5/18/00	20.0	0.03929	15.5	0.17274	18.5	0.16922	16.0	0.00000
6/21/00	22.0	0.09282	16.0	0.01819	23.0	0.31241	17.0	0.00002

[1]No discharge measurement taken due to technician error

Surface Water Chemistry

A discontinuous record of water quality exists for many of the sites within the Herrington Creek watershed since December of 1996. Sampling at all stations except HRTB was terminated in January 1998 and then resumed in December 1998 with the addition of other sites to better characterize the watershed. To better demonstrate streamflow conditions at each sampling occasion, the mean daily continuous flow record is depicted with each figure for the HRTB station, which is equipped with a recording gage. The results for ANC are presented in Figure 4. ANC values at Murley A, Bull, and the Upper Bull Glade Run site were consistently below zero throughout most of the sampling record, indicating chronically acidic conditions in these streams. During periods of low flow during the summers of 1997 and 2000 (which could be characterized as "normal" years for precipitation), ANC values at Murley A and Bull remained below zero. During the droughty conditions that existed during the summer and fall of 1999, however, ANC at the Bull site eventually increased to about 50 µeq/L; ANC remained in the

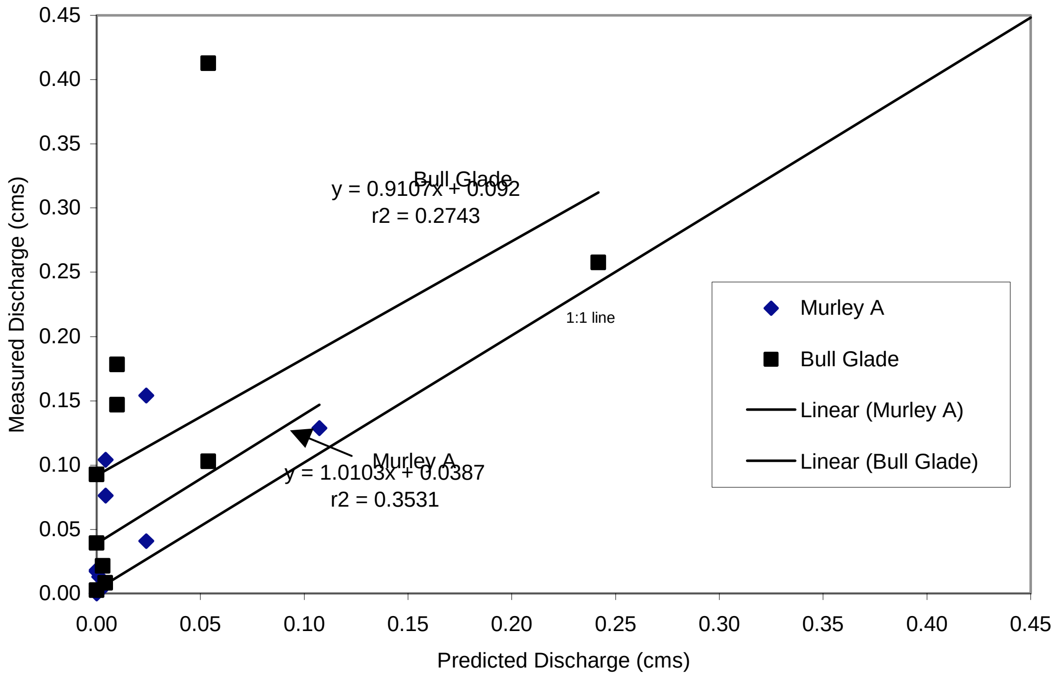

Figure 2. Relationships between predicted and measured discharge for the Murley Run (Murley A) and Bull Glade Run (Bull) stations.

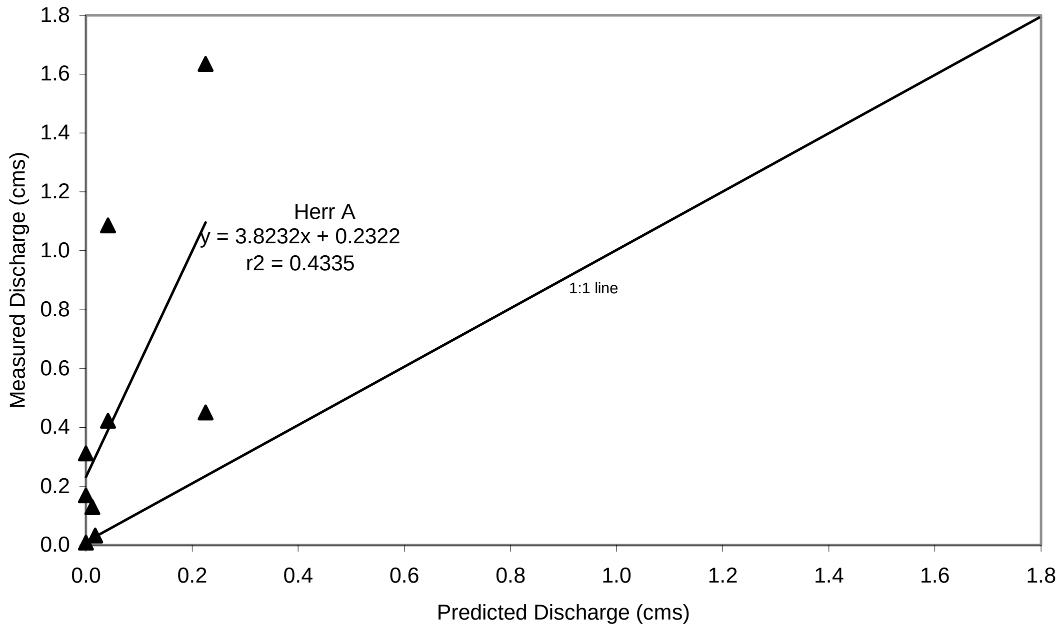

Figure 3. Relationship between predicted and measured stream discharge for the Herrington Creek (Herr A) station.

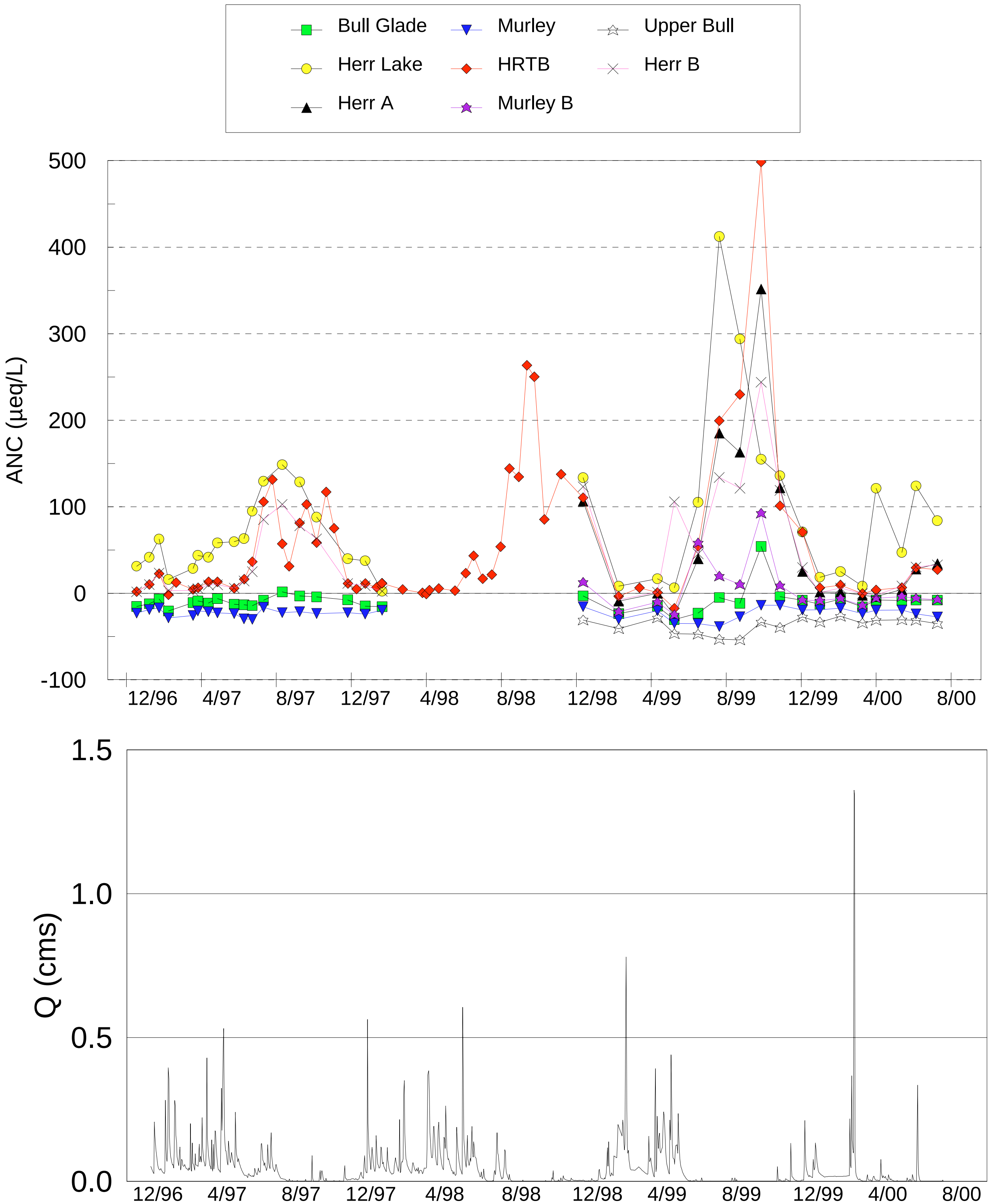

Figure 4. Temporal variations in streamwater ANC at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

negative range at the Upper Bull Glade Run site, however. ANC in Herrington Creek tends to exhibit more of an episodic response. During periods of low flow in normal and dry years, the ANC becomes quite high. When flows increase, ANC drops near or below zero.

Results for closed pH at the Herrington Creek watershed sites exhibited similar patterns as the ANC results (Figure 5). Closed pH and discharge exhibited an inverse relationship, with the lowest pH values at the Murley A, Bull, and Upper Bull stations. These values were typically lower than 5. Measurements in the Murley/Bull Glade system ranged between 4.1 at Upper Bull in December 1998 to about 6.3 at the Murley B and Bull sites in fall 1999 (Figure 5). Results confirm that current pH levels in both Bull Glade and Murley Runs would not support fish populations. Among all of the streams sampled, the outflow from Herrington Lake (Herr Lake) exhibited the overall highest pH values.

Results for both total reactive and exchangeable reactive aluminum also suggest that during most of the year, Murley and Bull Glade Runs could not support fish populations (Figures 6 and 7). Chronic aluminum levels are typically greater than 200 ppb, which is the toxicity threshold for most fish species. During the summers of 1997 and 1999, the aluminum concentrations did drop to acceptable levels at the lower sampling points in the watershed (Bull, Murley A, and Murley B). At the other sampling locations, total and exchangeable reactive aluminum concentrations tended to be less than 200 ppb, except during periods of higher flow.

Nitrate-N concentrations at all of the sampling stations were typically less than 0.5 mg/L throughout the sampling period (Figure 8). Nitrate concentrations were lowest at most of the sites during the most recent sampling in June 2000. Sulfate tended to follow a seasonal pattern, with concentrations tending to be lower in the summer months during low flow (Figure 9). It is interesting to point out the spikes in nitrate and sulfate concentrations in December following the

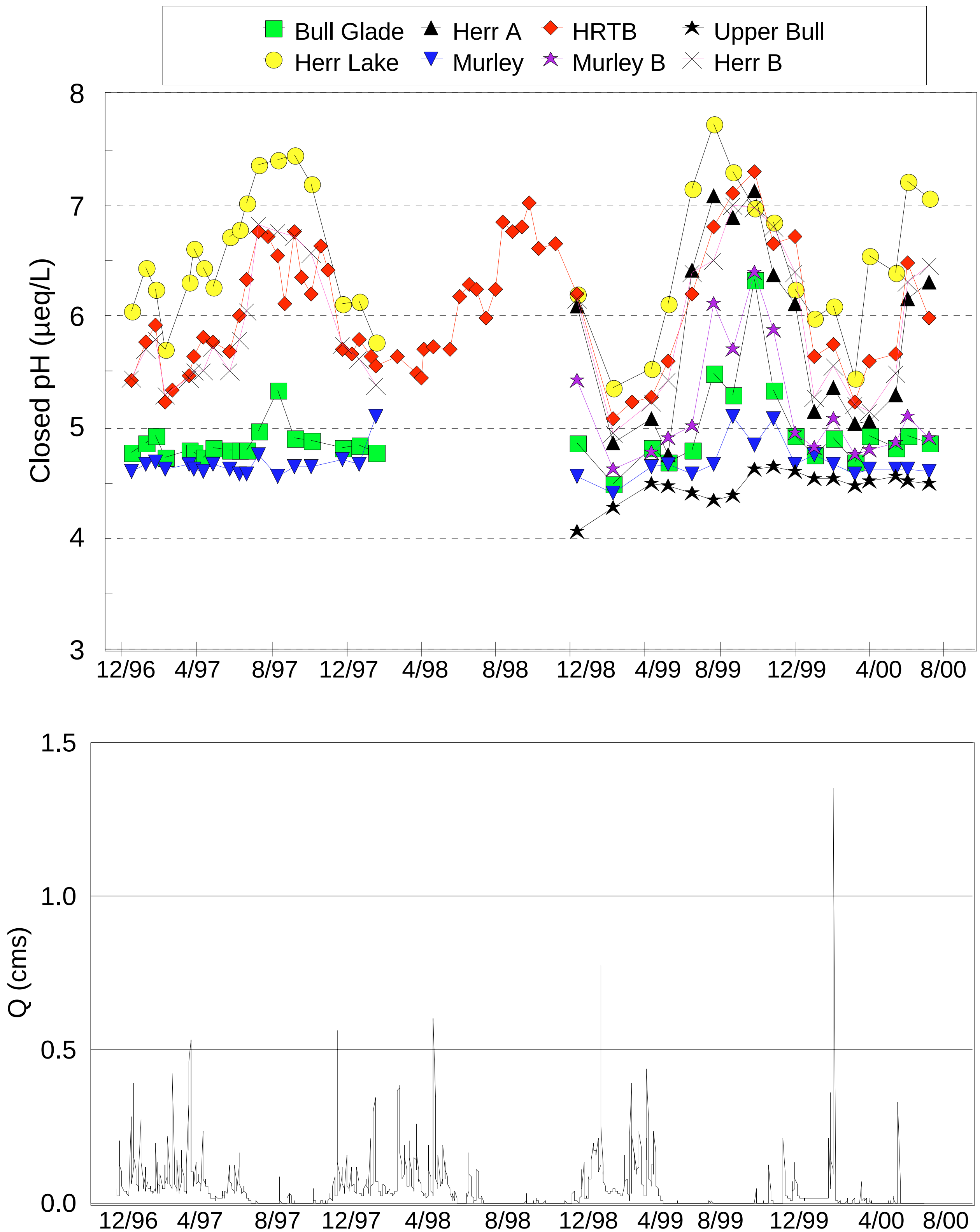

Figure 5. Temporal variations in streamwater pH (closed system) at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

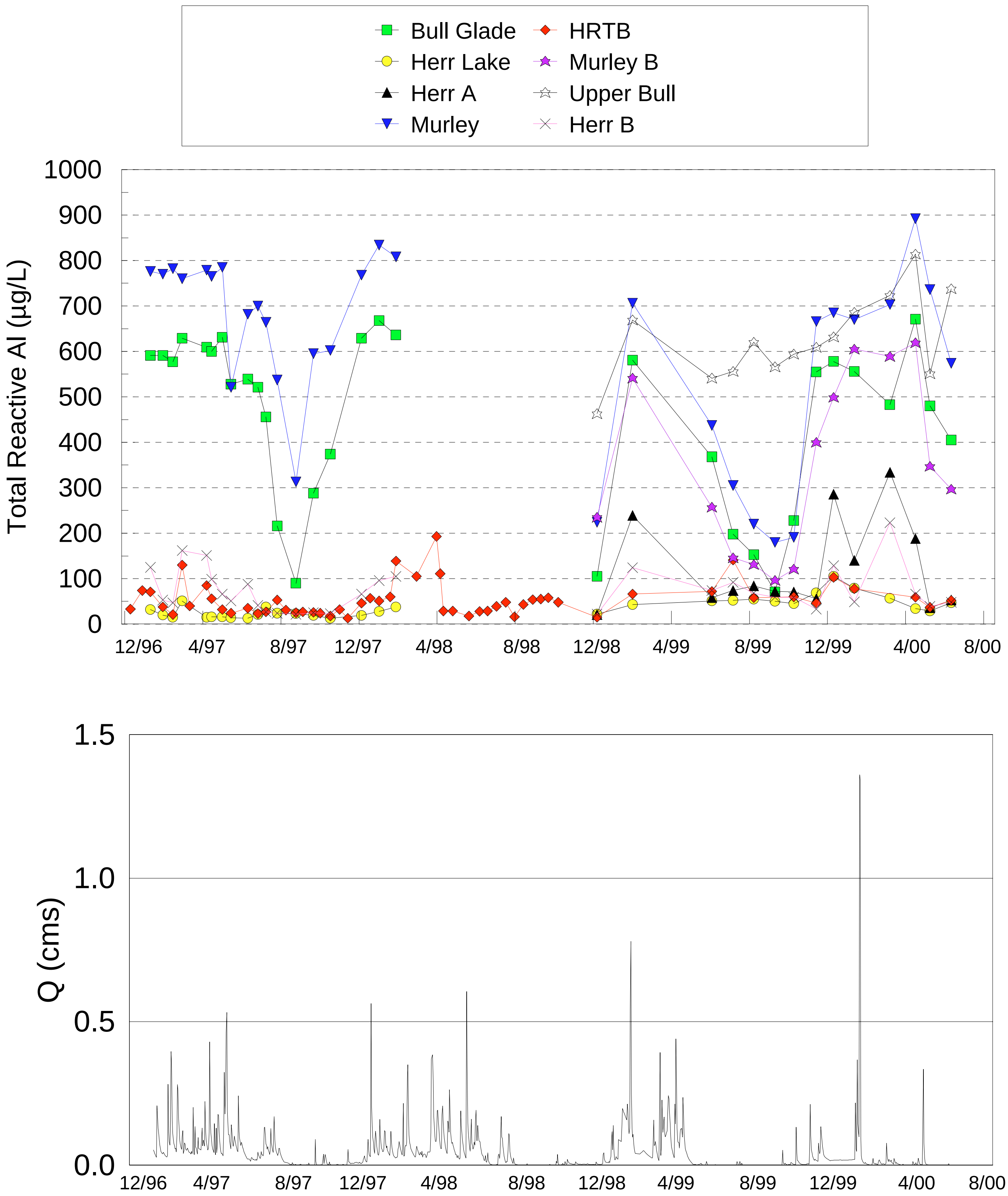

Figure 6. Temporal variations in streamwater total reactive aluminum concentration at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

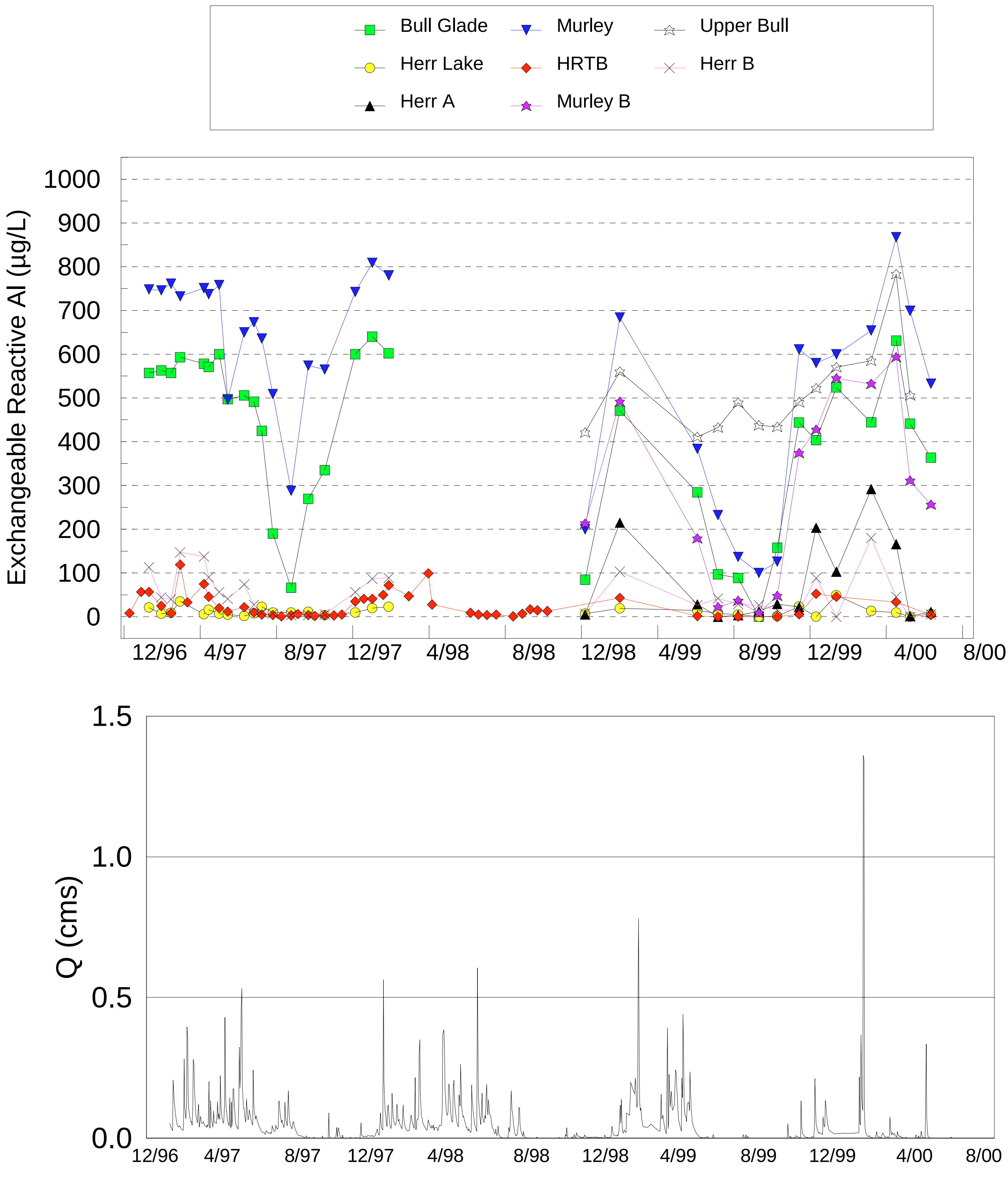

Figure 7. Temporal variations in streamwater exchangeable reactive aluminum concentration at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

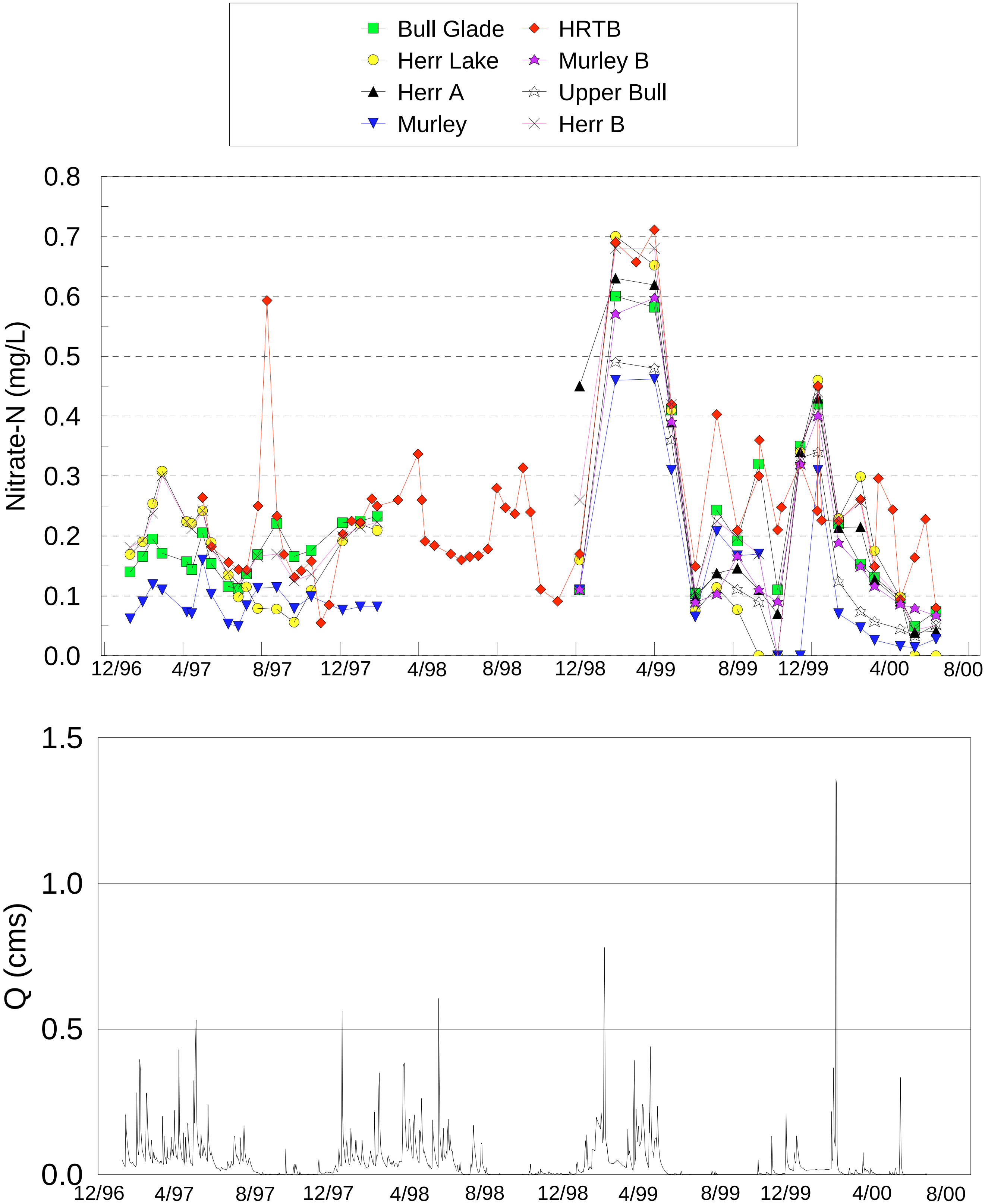

Figure 8. Temporal variations in streamwater nitrate-N concentration at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

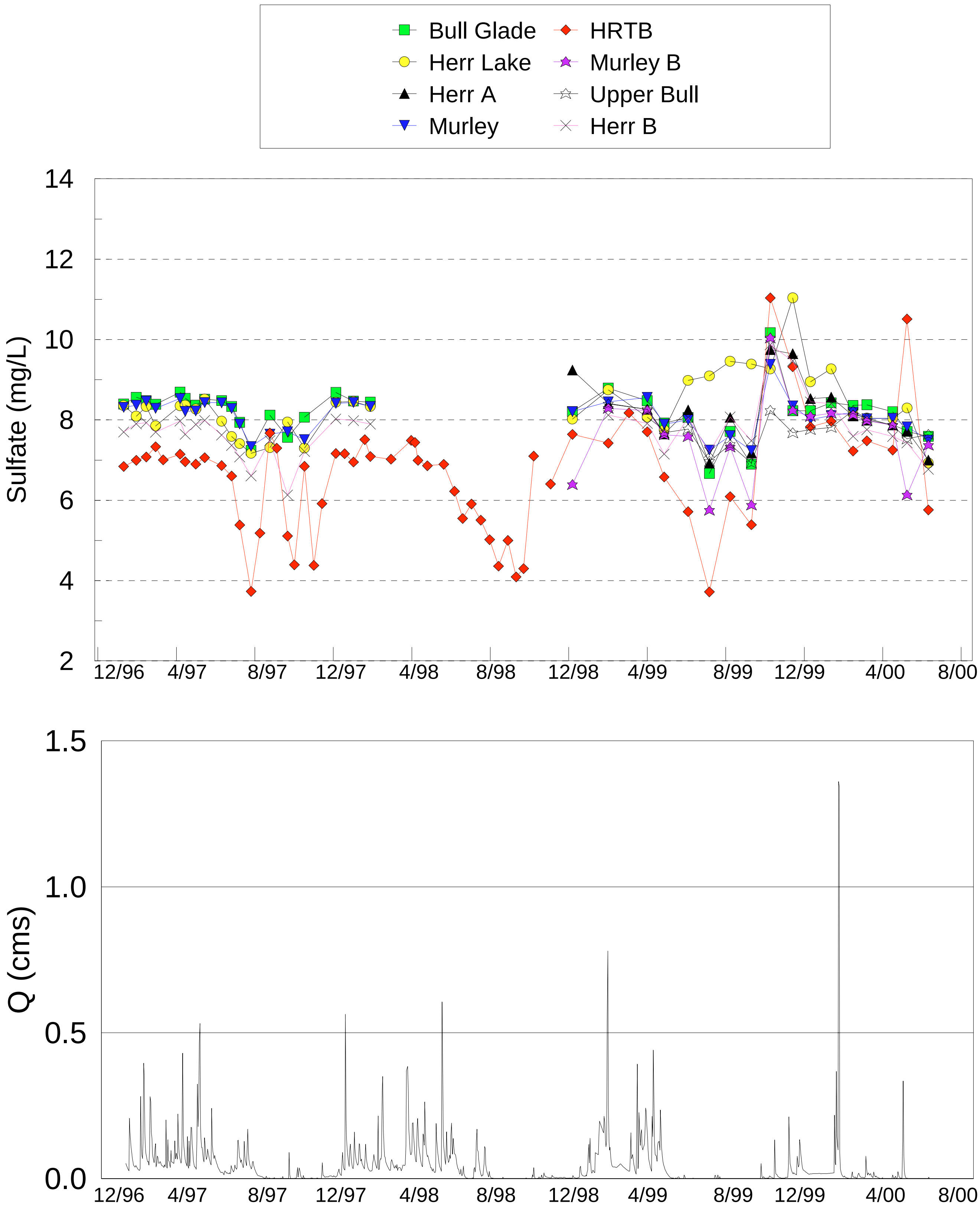

Figure 9. Temporal variations in streamwater sulfate concentration at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

drought in the summer and fall of 1999, perhaps suggesting that atmospherically-deposited pollutants depositedon the soil and forest canopy by dry deposition in the summer were flushed out in the early winter.

Base cation results verify the lack of buffering capacity in the Murley/Bull Glade Run system (Figures 10, 11). Streams with substantial buffering abilities typically have calcium concentrations greater than 4 mg/L and magnesium concentrations greater than 2 mg/L. Calcium concentrations at all of the sites tended to be higher during periods of lower flow, corresponding to increased ANC values.

Results from water quality monitoring also indicate that natural organic acidity does not contribute substantially to the acidity of streams within Murley and Bull Glade Runs. DOC was most often less than 2 mg/L at most of the upper sites, except during the fall of 1999 (Figure 12). These higher DOC's were more likely the result of the drought than from natural organic acidity, especially since DOC at all sites was highest during this period. There were substantial increases in DOC at HRTB, Herr A, Herr B, and Herr Lake stations during summer low flow periods.

The potential for precipitation of dissolved iron in the treated stream has also been considered to be a potential adverse effect of remediation of Murley or Bull Glade Runs. Iron and manganese were measured during the second water quality monitoring period. Iron concentrations were actually lowest at the upper reaches within the Herrington Run watershed, with the Upper Bull, Bull, and Murley A sampling sites experiencing iron concentrations typically less than 250 μg/L (Figure 13). Iron was highest at HRTB, Herr A, and Herr B stations, especially during periods of low flow. The U.S.E.P.A. has developed EcoTox thesholds (ET) for many metals that could have negative impacts on aquatic biota. ET's are defined as media-specific contaminant concentrations above which there is significant concern regarding adverse

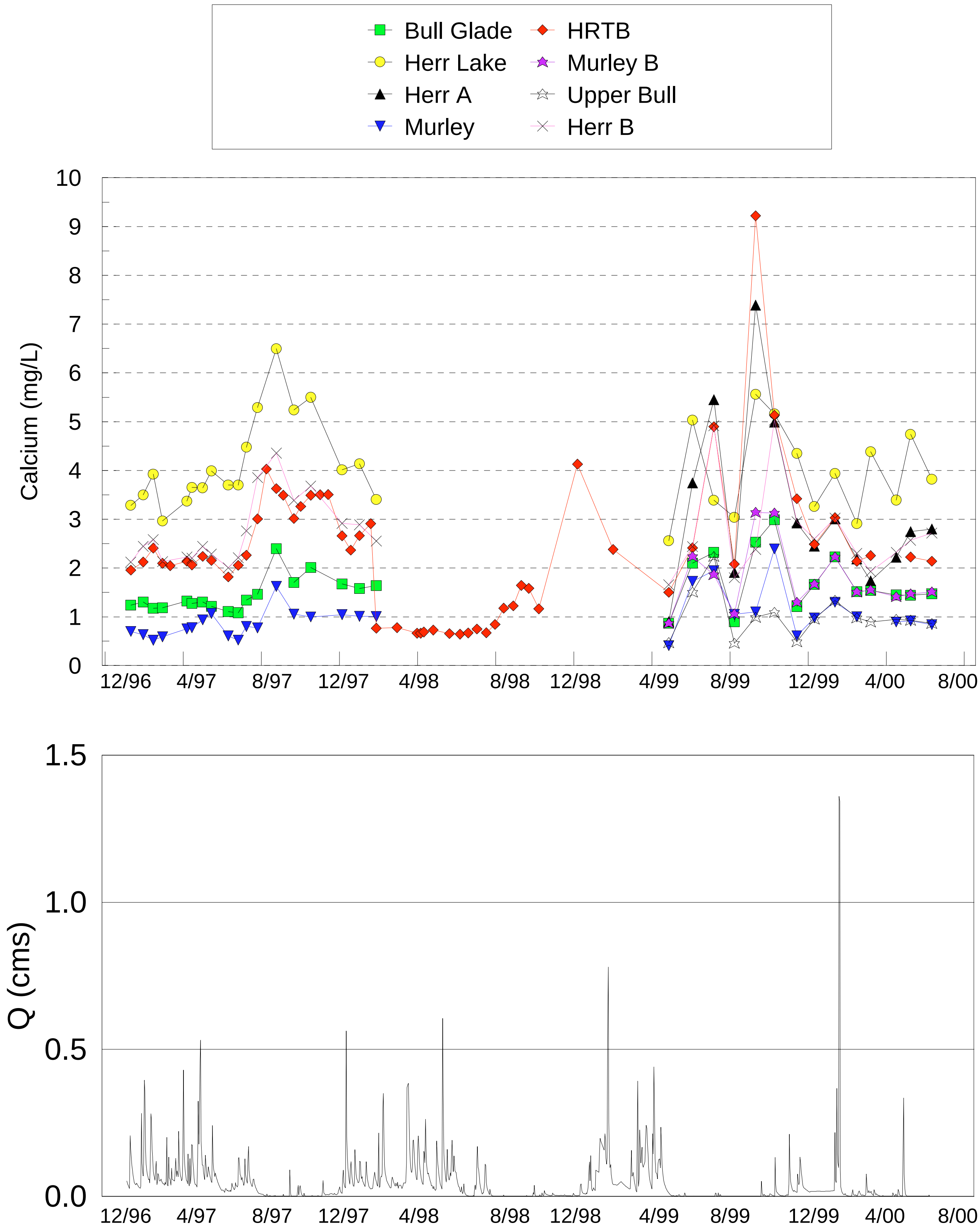

Figure 10. Temporal variations in streamwater calcium concentration at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

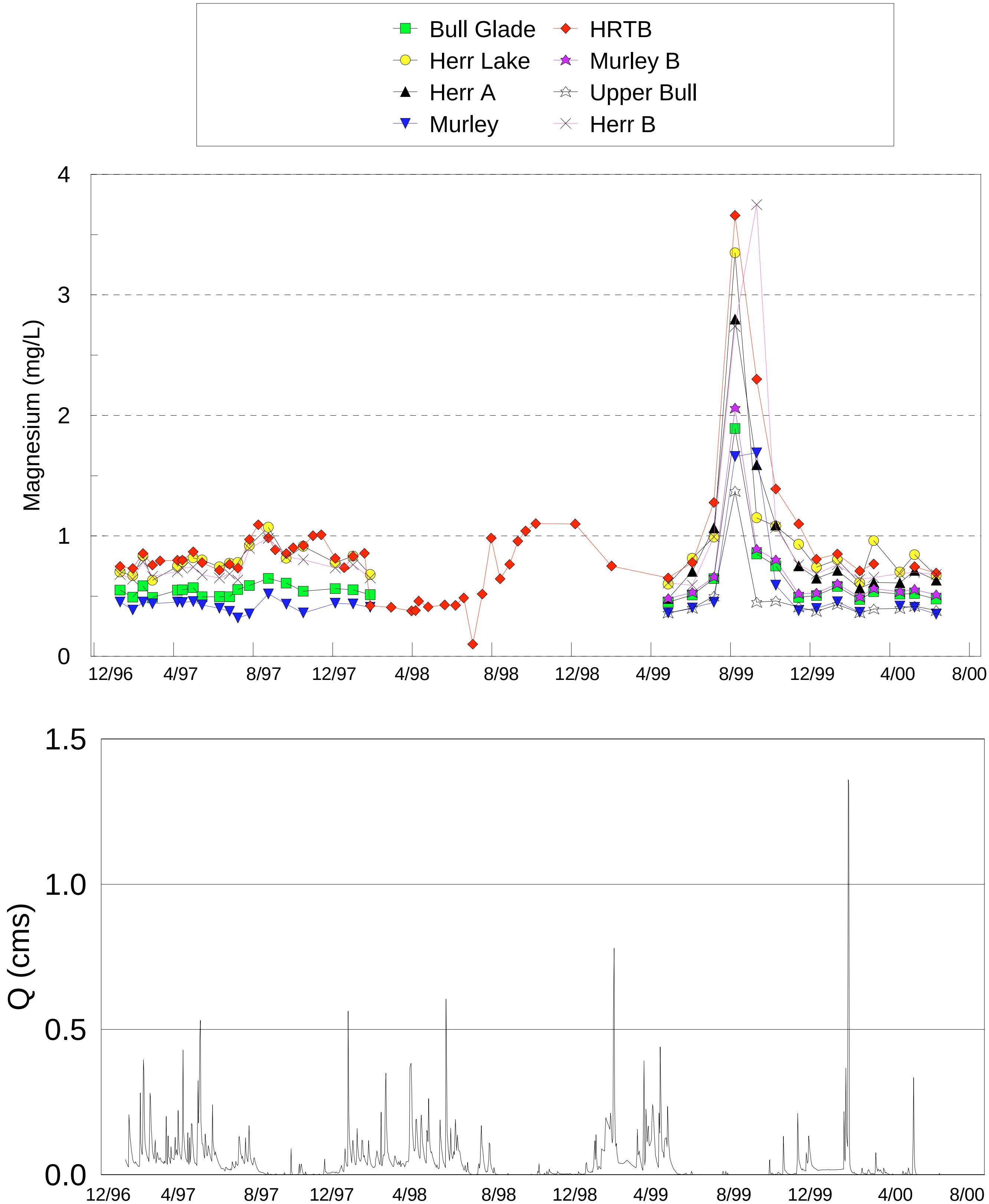

Figure 11. Temporal variations in streamwater magnesium concentration at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

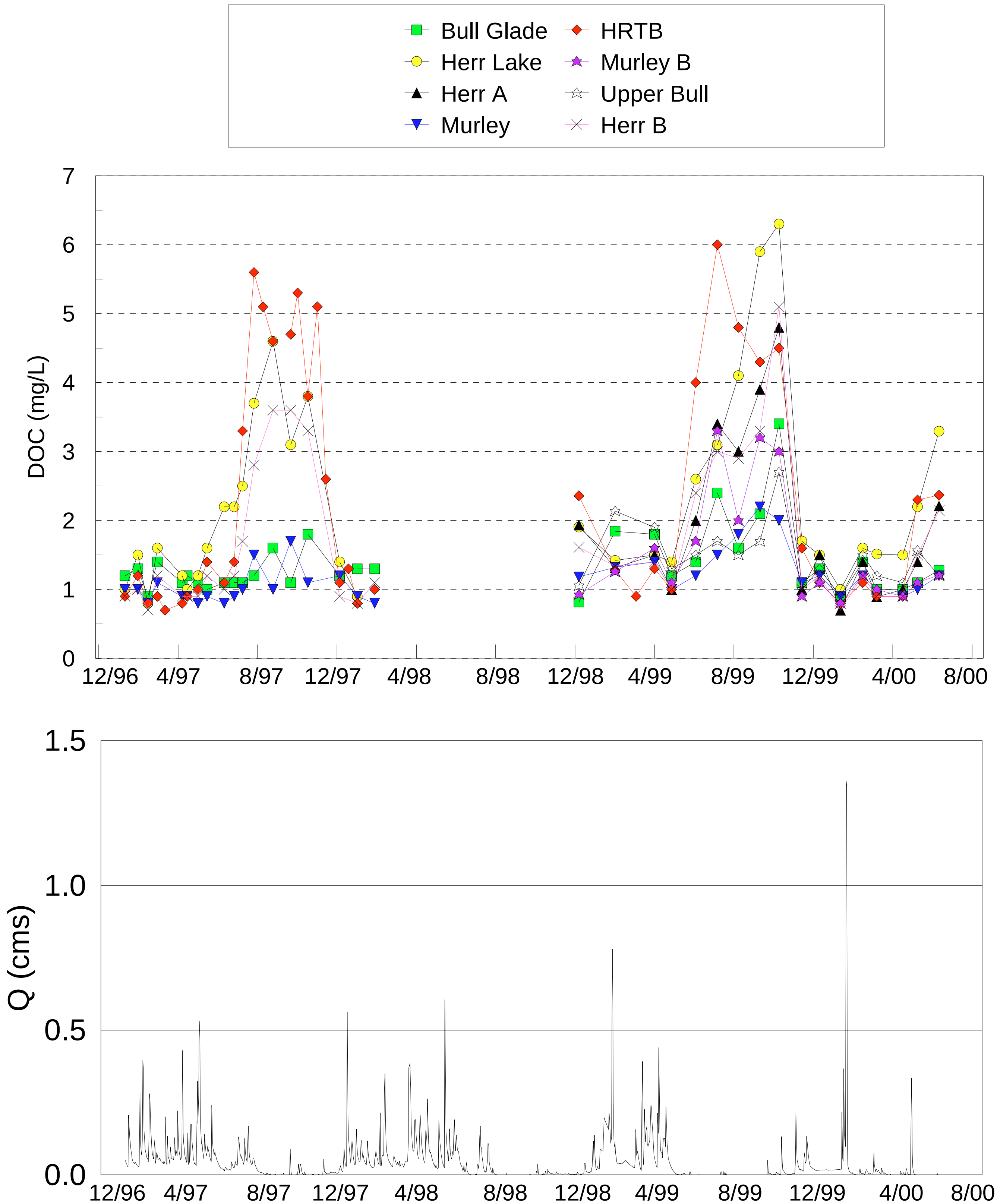

Figure 12. Temporal variations in streamwater DOC concentration at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

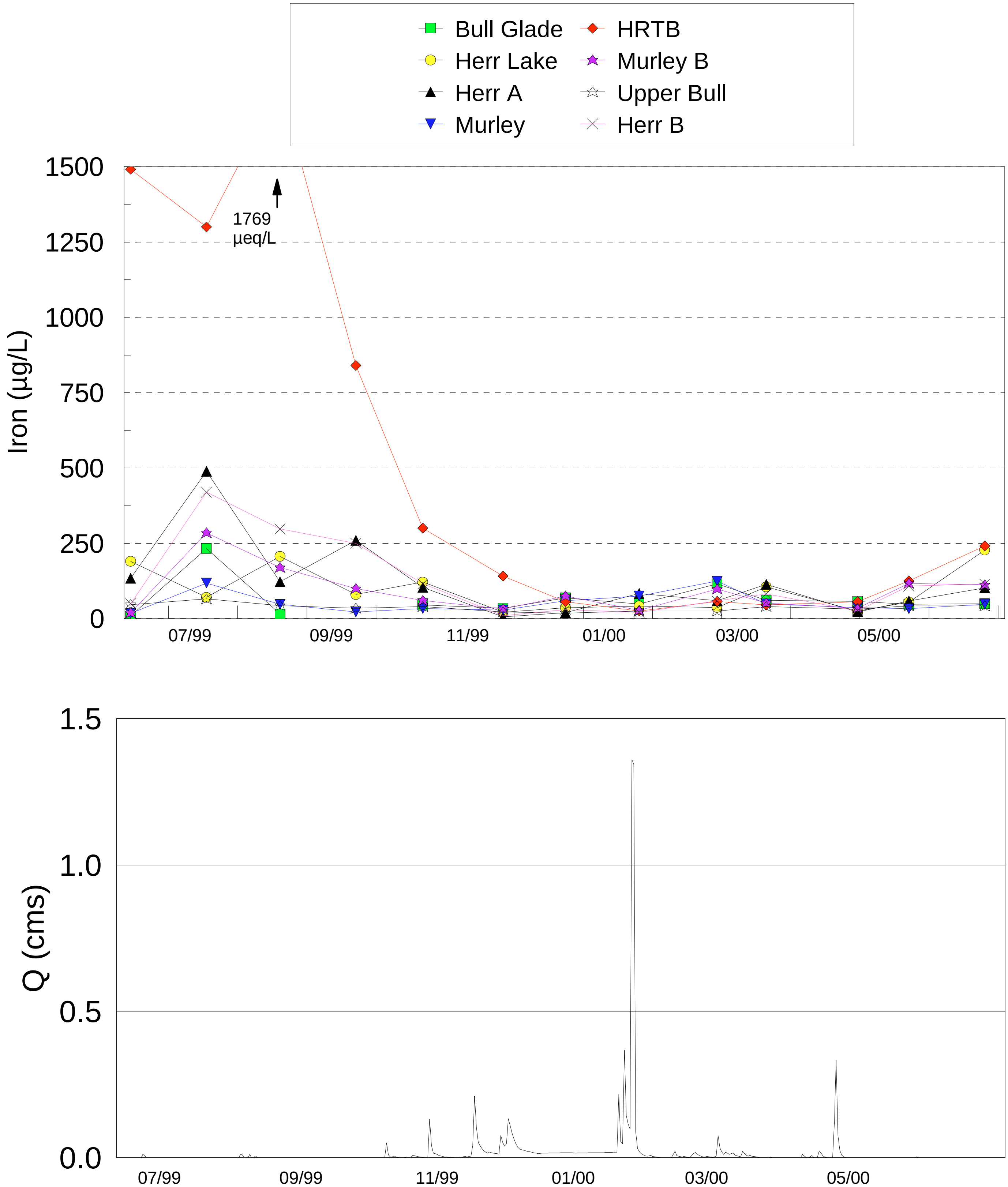

Figure 13. Temporal variations in streamwater iron concentration at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

ecological effects to warrant further site investigation. The ET's are for screening purposes and concentrations below the established benchmark values should not result in significant adverse effects to ecological receptors. Some of these adverse effects include reductions in survival, growth or reproduction of tested organisms. The ET's for surface water have been based on the Chronic Ambient Water Quality Criteria (CAWQC) developed by U.S.E.P.A. Office of Water. The ET for iron in surface waters has been determined to be 1000 µg/L. The only site that exhibited concentrations above this threshold was HRTB during the drought period of summer 1999.

Manganese was also measured during the sampling period (Figure 14). The ET for manganese is 80 µg/L. Manganese was highest overall at the Murley A site and most of the sites exhibited concentrations greater than the ET. Therefore, manganese levels should be closely monitored if any restoration project to improve water quality in this watershed is ever implemented.

In order to determine whether Murley or Bull Glade Runs are sensitive to the effects of episodic acidification, we attempted to sample these sites (and the Herr A site) during the critical spring snowmelt period of 2000. As a result of the drought conditions in the area during the summer of 1999 and winter 1999-2000, we were only able to sample one rainfall event in 2000—a modest rainfall event in early April (04/03/00 – 04/07/00). Samples were collected every 4 hours using American Sigma automated samplers. During the event, 0.67 inches of rain was recorded at Oakland, Maryland. The continuous water level recorder at HRTB noted an 11 cm rise in stream level over the sampling period, with peak flow occurring at approximately 3:00 a.m. on April 4th. The second event was a snowmelt episode sampled in February of 2001. Samples were collected every 6 hours and the continuous water level recorder at HRTB noted

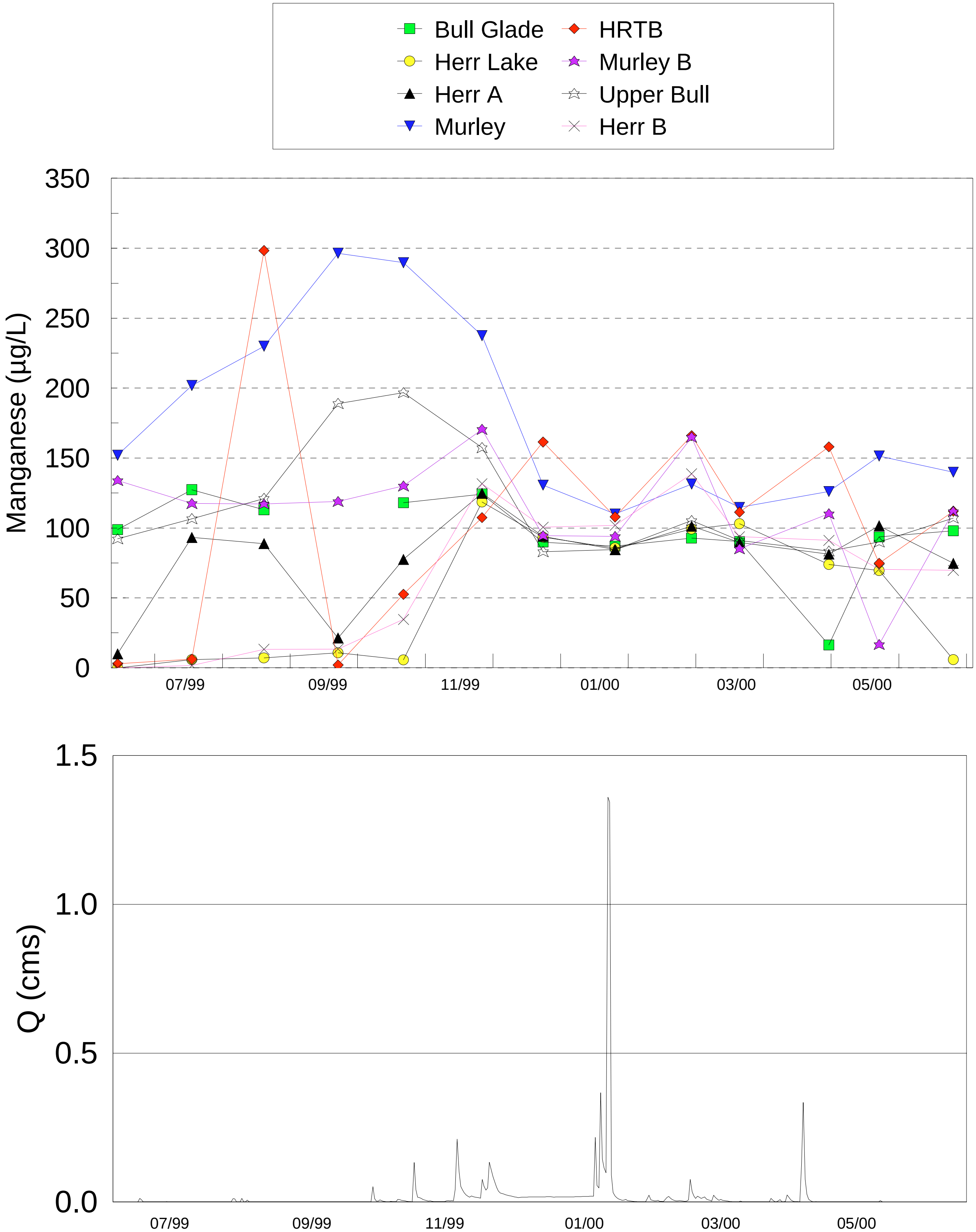

Figure 14. Temporal variations in streamwater manganese concentation at eight stations in the Herrington Creek watershed during the period 1/1997 through 6/2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

about a 20 cm rise in stream level over the sampling period. Although this event was only of a slightly greater magnitude than the 2000 event, it should indicate the response of these streams to snowmelt conditions.

Water chemistry results for pH, ANC, and exchangeable reactive aluminum for the April 2000 precipitation event are depicted in Figures 15-17. Peak flow occurred early on April 4th. Results suggest that the responses of Murley A and Bull to high flow events are not very dynamic. With the exception of one sample measured at the Bull site, ANC remained negative for the duration of the event, yet never dropped much lower than values typically observed during baseflow conditions. Indicative of the modest magnitude of this event, ANC at the Herr A site was also fairly stable during the sampling period and never dropped below zero (Figure 15). Open pH at both Bull and Murley A, with the exception of two Bull samples taken early in the event, remained between 5.0 and 4.5 and never exhibited large swings as are often experienced by episodically-affected streams. The pH values seen in these streams during the event closely reflected the pH values during baseflow as well (Figure 16). Exchangeable reactive aluminum concentrations during the April event were also found to be similar to baseflow aluminum concentrations. On almost all of the sampling occasions, aluminum was higher than the fish threshold toxicity level of 0.2 mg/L (Figure 17). Increases of about 0.2 mg/L were exhibited at both sites, suggesting contributions from soils within the watersheds.

Results from the 2001 snowmelt event suggest that the responses of Murley A and Bull to events are slightly more dynamic (Figures 18-20). During the 2001 event, Bull demonstrated an initial rise in ANC to almost 80 μeq/L before it dropped back below 0 μeq/L on the falling limb of the hydrograph (Figure 18). This phenomenon was not observed during the rainfall event in April of 2000. ANC at Murley A again remained negative throughout the snowmelt episode.

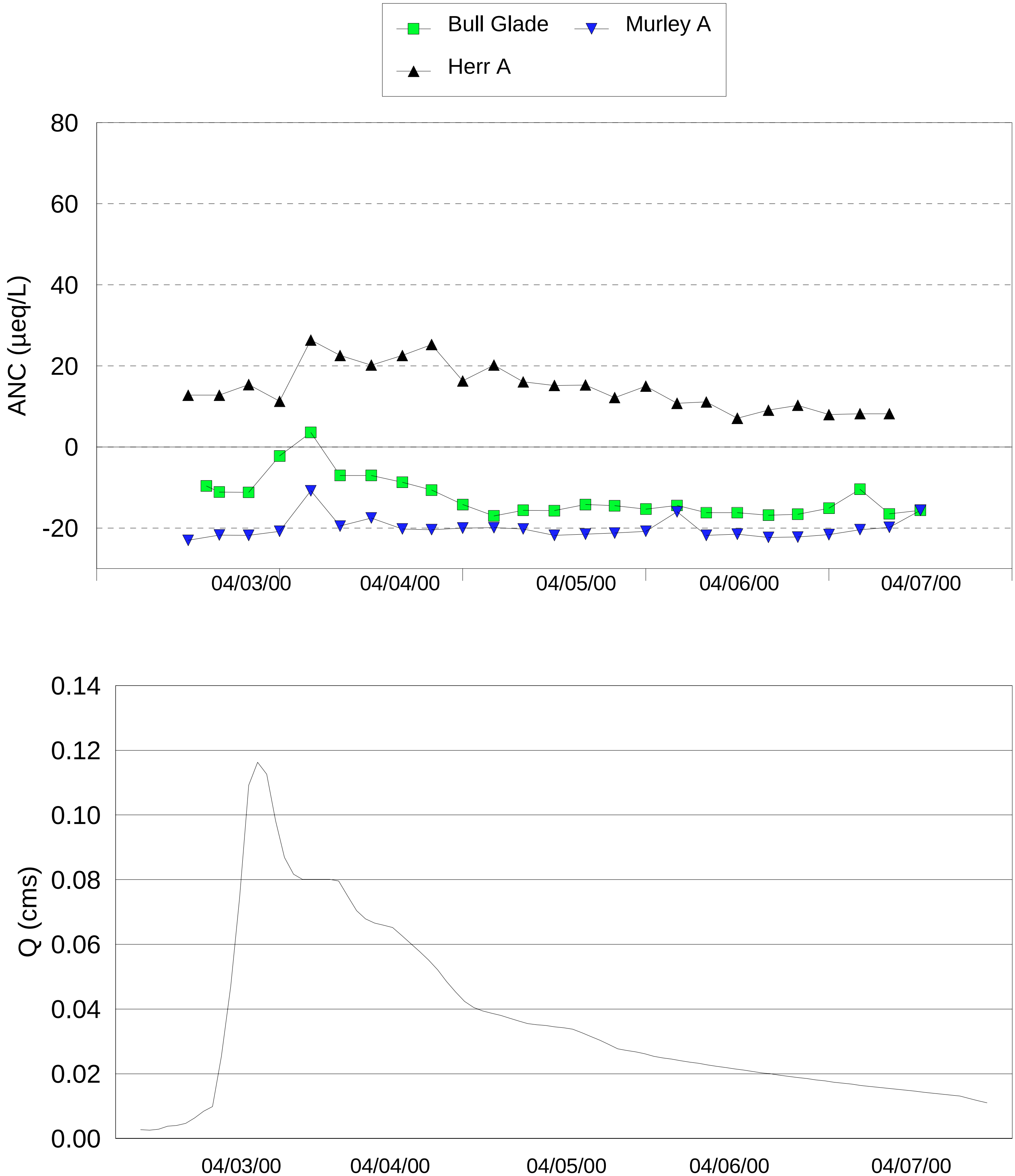

Figure 15. Temporal variations in streamwater ANC at three stations in the Herrington Creek watershed during the period April 3-7, 2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

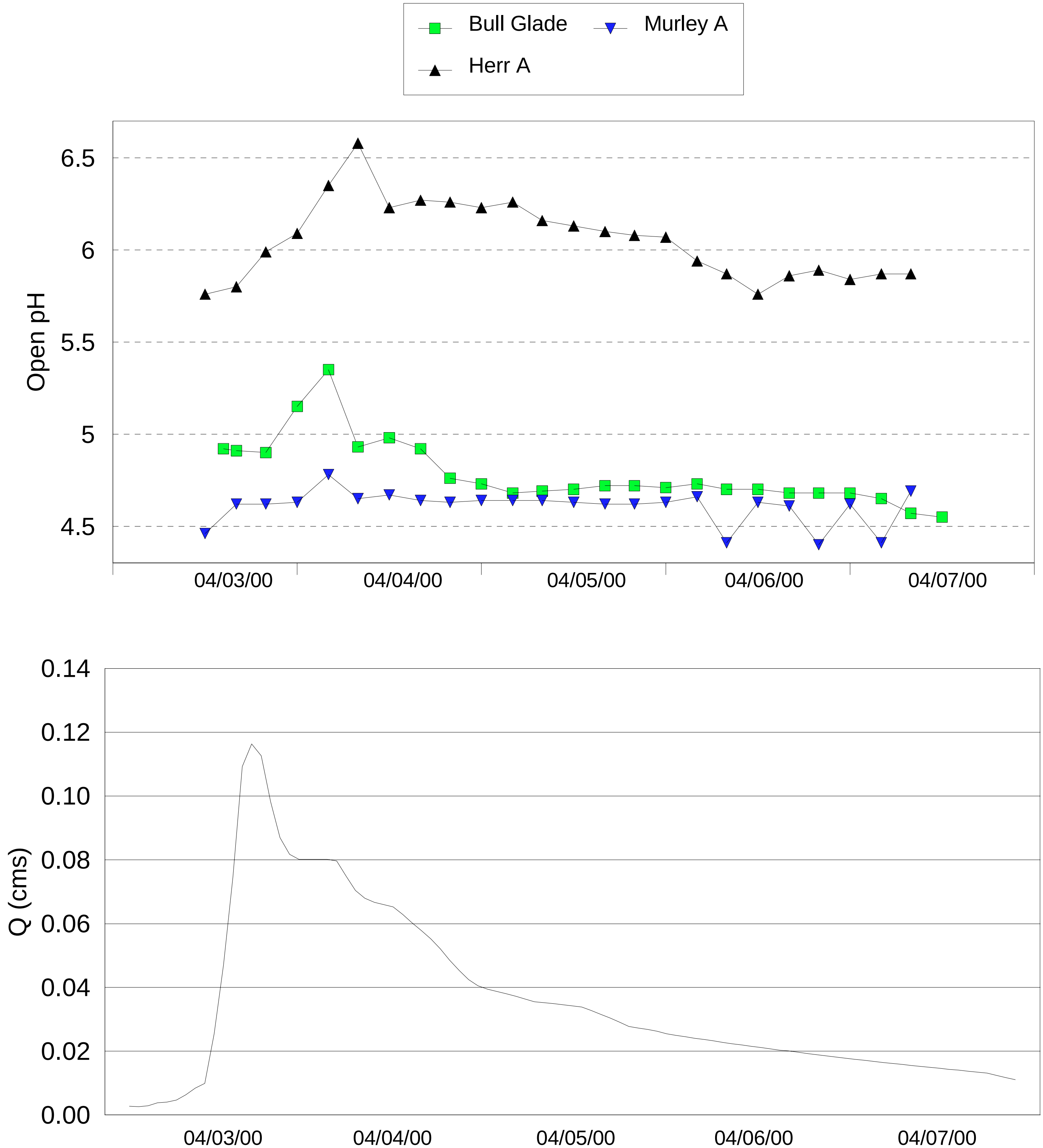

Figure 16. Temporal variations in streamwater pH (open system) at three stations in the Herrington Creek watershed during the period April 3-7, 2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

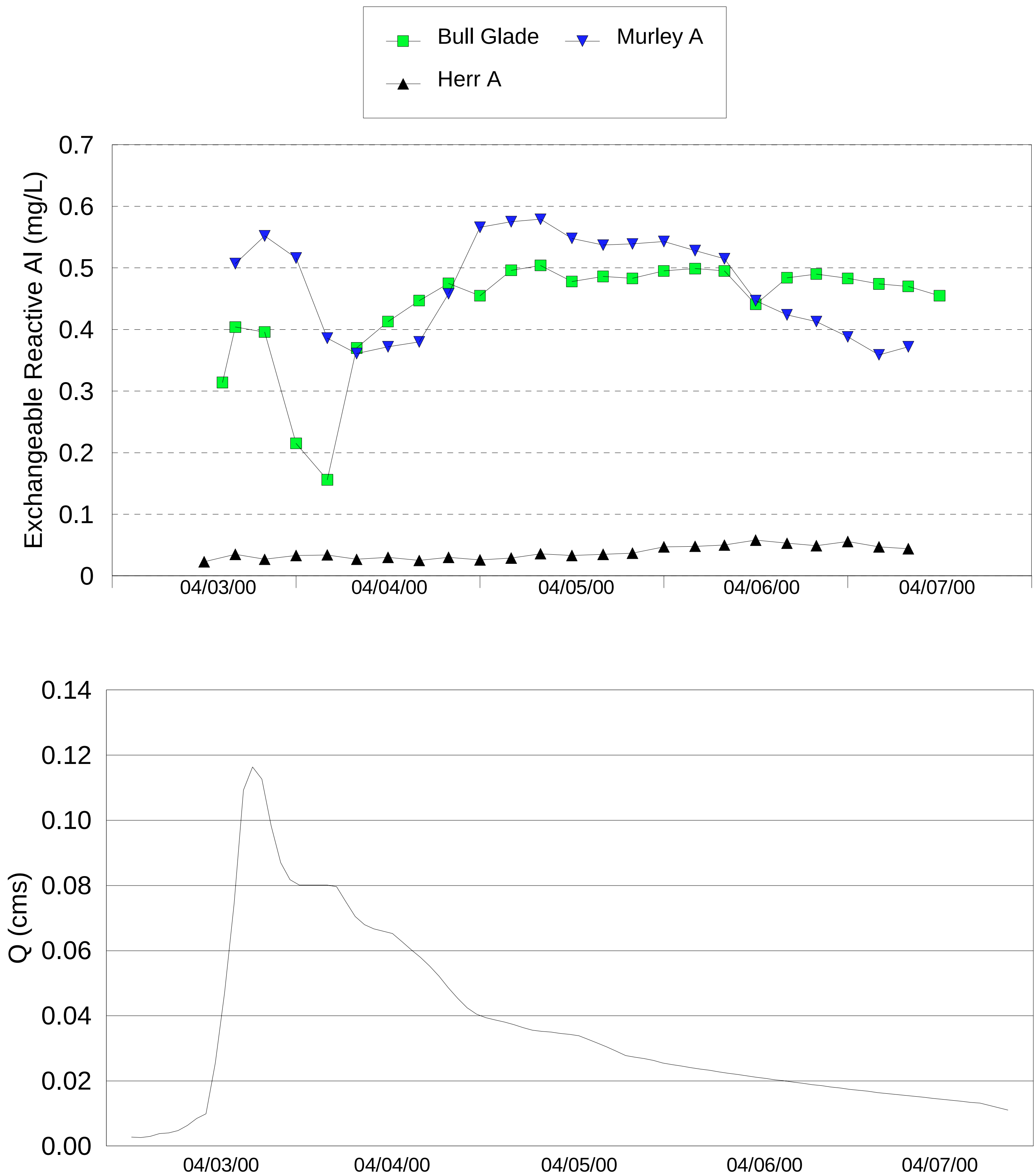

Figure 17. Temporal variations in streamwater exchangeable reactive aluminum concentration at three stations in the Herrington Creek watershed during the period April 3-7, 2000 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

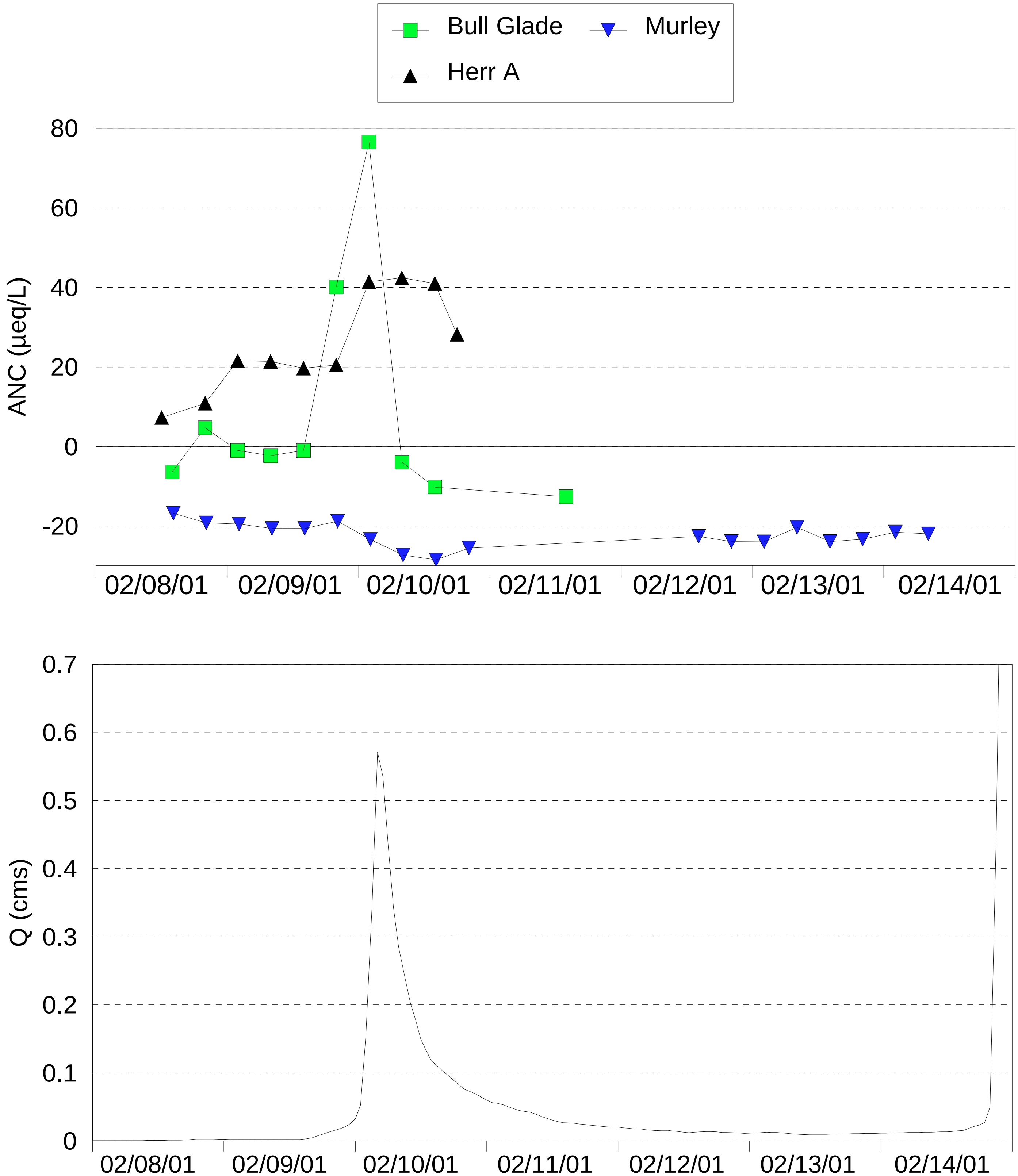

Figure 18. Temporal variations in streamwater ANC at three stations in the Herrington Creek watershed during the period February 8-14, 2001 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

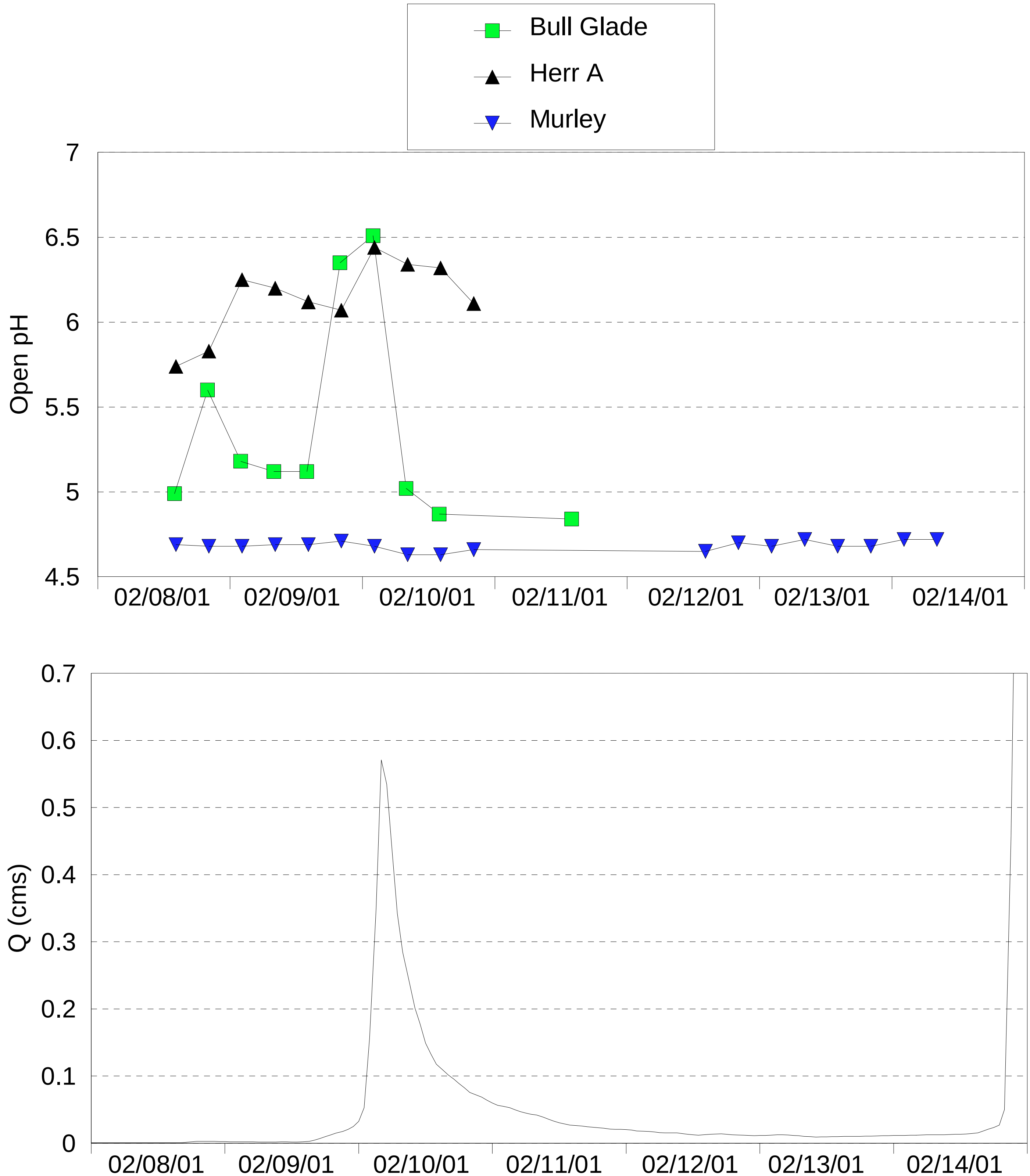

Figure 19. Temporal variations in streamwater pH (open system) at three stations in the Herrington Creek watershed during the period February 8-14, 2001 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

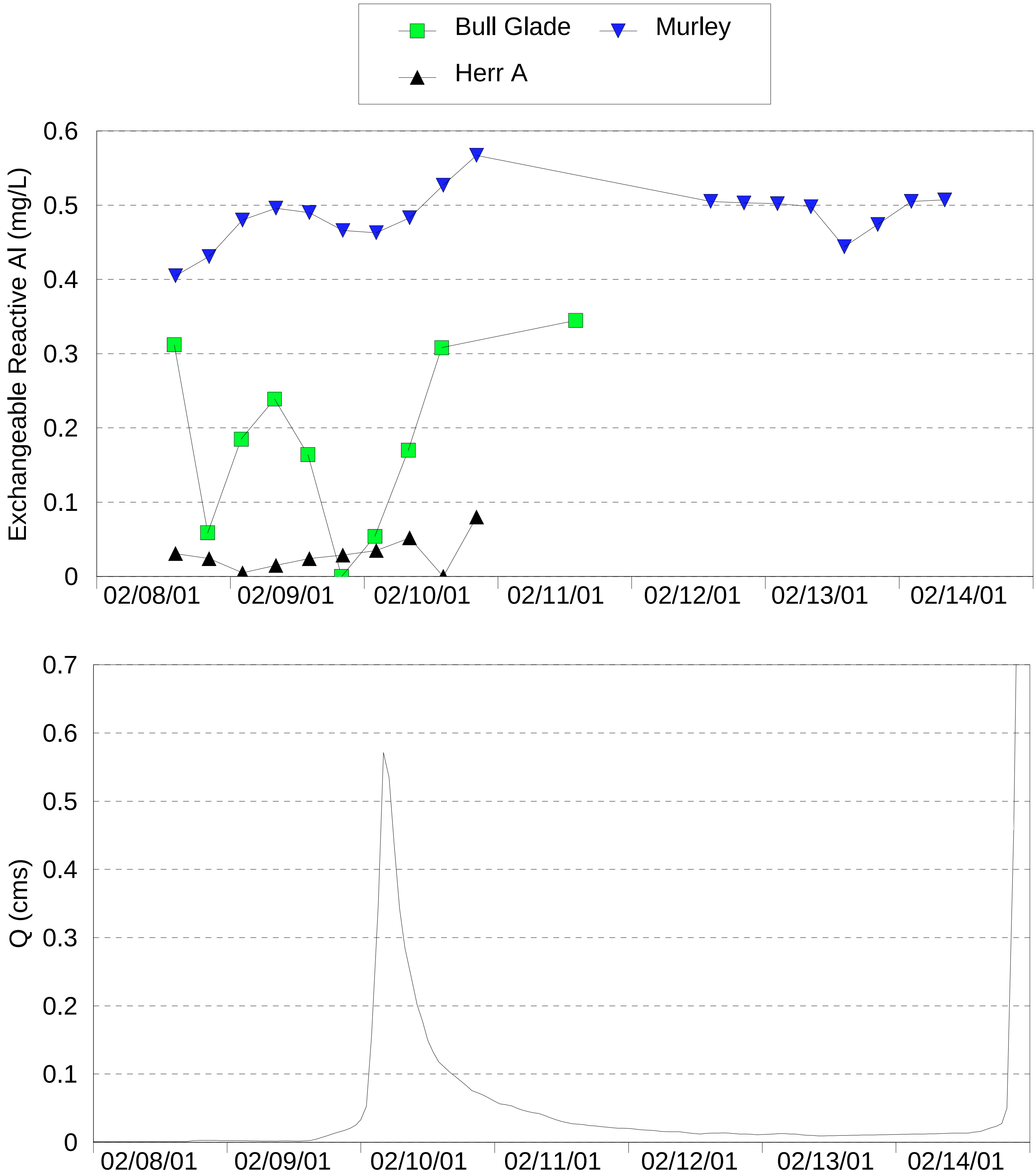

Figure 20. Temporal variations in streamwater exchangeable reactive aluminum concentration at three stations in the Herrington Creek watershed during the period February 8-14, 2001 (upper frame). Daily stream discharge (Q) at HRTB also shown (lower frame).

The response of Herr A also rose initially with at peak flow. Low temperatures, though, resulted in equipment failure at this site. Samples from the receding limb of the hydrograph were not collected after the sampler froze up. Therefore, it is unclear whether ANC at Herr A was negative later during the event. Open pH at Bull exhibited a rise in pH that corresponds to the increase in ANC observed at about peak flow (Figure 19). Otherwise, pH at the two possible treatment sites remained between 4.5 and 5.0 units. Exchangeable reactive aluminum concentrations during the snowmelt event were found to be similar to baseflow aluminum concentrations with the exception of the peak flow samples collected from Bull (Figure 20). At Murley A, exchangeable aluminum was higher than the fish threshold toxicity level of 0.2 mg/L. At Bull, non-exchangeable aluminum was higher than the toxicity threshold at the beginning of the event at lower flows and again on the receding limb of the hydrograph, suggesting that hydrologic pathways during events might be more complex at this site.

In conclusion, water quality data for the two-year period indicate that stream sulfate concentrations at both the Bull Glade and Murley Run sampling stations were consistently less than 200 µeq/L—much lower than sulfate values (>500 µeq/L) commonly associated with mining impacts. The average total acidity value measured by Gannett Fleming (1997) on Bull Glade Run influent water used for their column experiments was 7.6 mg/L, which is very low compared to values commonly measure in acid mine drainage (AMD) effluents (200 mg/L). Further, the low dissolved organic carbon (DOC) concentrations (<2 mg/L) within Murley and Bull Glade Runs indicate that natural organic acidity from wetlands does not significantly contribute to the acidity of these streams , thus ruling out the interpretation that the watershed is "naturally acidic". The bedrock geology and associated soils within the watershed are known to impart little buffering capacity to effluent streams (Janicki, 1990; Meagher, 1995) and this contribution has very likely been exhausted as a result of the long history of acidic deposition from the atmosphere. Results from the episodic samples suggest that Murley Run and Bull Glade do not experience any appreciable episodic acidification due to the extremely low chronic levels of ANC in these streams. We thus conclude that both Murley and Bull Glade Runs are chronically acidified and are not substantially influenced by episodic acidification—very similar to the response of HRTB that has been described in much greater detail (Eshleman *et al.*, 2000). Since the acid-base chemistry of these streams closely reflects rainfall chemistry under baseflow conditions, large changes in pH and ANC would not typically be expected under stormflow conditions.

Benthic Macroinvertebrates

Data from the benthic macroinvertebrate study are presented in Appendix A (Tables A1—A6). A list of the numbers of taxa collected from each station can be found in Table A1. Table A2 summarizes the functional feeding groups found in the CPOM samples at these stations. The shredder population was dominated primarily by *Leuctra sp.* and immature nemourid nymphs. The numbers of shredders were generally higher in Murley Run samples than in samples collected in spring from the unnamed tributary to Herrington Creek (HRTB).

Table A3 summarizes the metrics that were calculated for the aquatic D-net samples collected at each Murley Run station. A low order, minimally impacted Maryland Biological Stream Survey (MBSS) reference station on Little Bear Creek in the Youghiogeheny River drainage was selected for comparison purposes. Calculated metrics from benthic macroinvertebrate collections made at this site in 2000 are also presented in Table A3. Because it is an MBSS reference station, it is considered to be representative of non-impacted streams in the drainage basin.

Mean taxa richness, which indicates stream health, was approximately 18 at the Murley stations, while a taxa richness of 22 was observed at Little Bear Creek. Taxa richness typically increases with improving water quality, habitat diversity, and habitat suitability. Taxa richness values greater than 22 are considered to be high, suggesting that taxa richness is somewhat low at the Murley stations. It is also important to point out that Plecoperteran nymphs accounted for well over 50% of the benthic macroinvertebrates collected at the Murley stations.

An EPT value greater than 12 would generally be considered high. EPT taxa ranged from 7 to 9 taxa at the Murley stations. These three orders of aquatic insects are considered to be pollution sensitive. Therefore, streams with higher EPT values are generally considered to be less impacted. The EPT taxa value from the reference station was 16, which is significantly higher than the EPT taxa observed at the Murley stations. Few ephemeropteran or tricopteran larvae were found at any of the stations in Murley Run, resulting in low total ephemeropteran taxa and percent Ephemeroptera values. These can be indicative of both organic pollution or acidification effects. There presence of stresses will reduce the abundance of mayfly, or ephemeropteran, groups.

The dipteran taxa metric can indicate good water and habitat quality. Generally, stations with higher taxa richness had greater numbers of dipteran taxa present.

The percent Tanytarsini was low at most stations. Although the number of intolerant taxa seen at most of the stations was fair, many taxa were represented by only a few individuals. A higher percentage of Tanytarsini can indicate lower levels of anthropogenic stress. Percent Tanytarsini was comparable between the Murley stations and the reference site.

The number of intolerant taxa seen at most of the stations was fair. Many taxa, though, were represented by only a few individuals. Little Bear Creek exhibited the highest intolerant taxa rating. Intolerant taxa typically decrease as more tolerant, opportunistic taxa increase.

Percent tolerant taxa values tend to increase as perturbation increases. Percent tolerant taxa are individuals with a tolerance rating of 7 to 10. Tolerant taxa include oligochaete, simulidae, and many chironomid larvae. Higher percent tolerant taxa values were seen at most of the Murley stations. Only two of the lower stations exhibited percent tolerant taxa values lower than the value observed at the reference station on Little Bear Creek. Large numbers of simulid larvae contributed to the high tolerant taxa value seen at stations 5 and 7.

The percent of collector gatherers, which typically decreases with increasing stress, was low (less than 13.5%) at the Murley stations. Conversely, percent collector gatherers observed at the Little Bear Creek reference station was 35%, suggesting that the Murley watershed stations experience higher stress levels than the reference station on Little Bear Creek.

IBI values for the Murley stations ranged from 1.89, which is considered to be very poor, to 3.00, which is considered to be fair. Stations 2, 3, and 15 demonstrated fair IBI values. Station 9 had the lowest IBI value. The two wetland areas in the watershed above this station could have influenced the IBI value. The lack of ephemeropteran nymphs, a common collector gatherer that is sensitive to acidic stream conditions, may have contributed to these low metric values at the Murley watershed sites. The IBI calculated for the reference station on Little Bear Creek was 4.11, which is higher than all of the Murley sites, and can be considered to indicate good benthic conditions. Increases in stream pH through stream restoration in the Murley Run watershed would be expected to result in higher IBI values. IBI values of 4.00 or higher, as seen in the reference station, would be anticipated.

A list of the taxa collected in the “T” samples at each station can be found in Table A4. Hill’s diversity measures are summarized in Table A5. Mean number of taxa varied from 3 at some stations to 8 at others with a mean abundance of 8 to 66 individuals collected. At most stations, a single taxon dominated the sample by 50% or more. These densities will provide a baseline from which comparisons can be made for future collections.

Overall, the benthic community in the Murley Run watershed does not appear to be entirely depauperate. Taxa richness scores of 16-22 are in the moderate range, which seems to be where many of the Murley stations fall. EPT taxa and intolerant taxa, however, are lower than values seen at Little Bear Creek. If acid mitigation action were to proceed, we would expect to see an increase in taxonomic diversity at the treated sites. Increased numbers of ephemeropteran and dipteran taxa, especially, should be expected.

Assessing Possible Mitigation Techniques

Alkaline Groundwater Pumping

We examined the area of Murley Run and Bull Glade Run for a site that would be accessible by vehicle and experience minimal/no loss of trees during installation of electrical power and would have a stream bed and flow adequate for injection and mixing of groundwater. We found such a site in the upper reaches of watershed, adjacent to Bull Glade Run (Figure 1). Here, the stream is approximately 15 m west of Snaggy Mountain Road and approximately 0.6 km south of the intersection of Snaggy Mountain and Cranesville Roads. A remnant logging road in the area could provide vehicular access to the stream's edge with little or no loss of trees. The stream bed is slightly incised at this location and experiences flow even during drought conditions like those experienced during the summer and fall of 1999.

The closest electrical power is located near the Garrett State Forest boundary (Figure 1), about 2.2 km north of the proposed well site location adjacent to Bull Glade Run. In an effort to keep environmental and visual disturbances in the state forest lands to an absolute minimum, we examined underground power lines and distributed sources of power only. A contractor provided the project with a cost estimate to install a private underground electric line from the last known electrical pole to the proposed well site (approximately 1.35 miles of line) where AC power would be needed to operate a submersible well pump (< 5 hp), as well as lights and electrical equipment in a small shed. Penn Line Services, Inc. estimated that it would cost approximately $54,000 to install the underground electrical service and all appurtenances at the proposed site. It would cost an additional $4,000 to install a 2” PVC conduit to house telecommunication lines for remote access to the pump house. Both the State Park Superintendent and the Garrett County Engineer gave the contractor preliminary verbal approval to install the trench line either on the road shoulder or in the ditch line adjacent to the road to minimize adverse impact to the surrounding forest and streams. In addition, Allegheny Power estimates it will cost $300 to $350 per month to run the 5 hp pump continuously.

Due to the rather high cost of installation of underground electric service to the proposed site, sources of distributed power were examined. Two viable options are: (1) a propane powered generator and (2) a fuel cell. Allegheny Energy Solutions supplied the project with a cost estimate for a generator capable of supplying adequate power to continuously operate a 5 hp pump motor that requires 220 voltage. A prime 25kW propane powered Generac generator would cost approximately $12,000. It would cost an additional $5,000 (approximate) for taxes, propane and installation (concrete pad, hook-up to breaker box, lighting and wiring). The generator is housed in a weather resistant case and is equipped with a muffler to minimize sound impacts. Cost of liquid propane to power the generator continuously (24 hour/day) would be approximately $350 per month. Perhaps the most environmentally-friendly energy source for use in a pristine area such as the Murley/Bull Glade watershed is the fuel cell. Essentially, a fuel cell converts hydrogen (from the processing of propane, natural gas, methanol or petroleum) to electricity without combustion. Heat and water vapor are the only by-products from the fuel cells electrochemical reaction. Although not commercially available at the present time, several companies report they are in the process of testing the fuel cell for use as residential or small commercial power plants. Neither of the two companies contacted directly would speculate on a release date or provide project personnel with an estimated cost, however. If the need for electrical power at this site comes to fruition within the next few years, fuel cells should be seriously considered as the power source due to their minimal environmental impact.

The Murley and Bull Glade watersheds are located within the Appalachian Plateau physiographic province of Maryland (Amsden, 1953; Vocke and Edwards, 1974). In general, the bedrock of this region is made of gently folded shale, siltstone and sandstone. The proposed well site is located on the eastern edge of the Upper Youghiogheny synclinal basin where the geologic formations are defined in terms of the persistant coal beds (Figure 21). According to Amsden (1953), the Conemaugh formation is the uppermost geologic formation exposed at this location. The Conemaugh is underlain by the Allegheny and Pottsville formations (Figure 22). The complete Conemaugh formation is 825 to 925 feet thick and is comprised primarily of gray/brown claystone, shale, siltstone and sandstone. However, in the Upper Youghiogheny basin, over half of the thickness has been removed by erosion, leaving the lower most strata whose distinctive members are black shales with thin beds of fossiliferous limestone, red clays and claystone with interbedded freshwater limestones. This lower member of the Conemaugh formation is bounded by the Barton Coal bed, located mid-formation in the Conemaugh, and the Upper Freeport Coal bed, located at the top of the Allegheny formation. The Allegheny formation is 300 feet thick and is characterized by massive sandstone beds (fine grain size to conglomerates) and numerous coal beds of varying thickness. These coal beds are typically underlain by underclays and argillaceous freshwater limestones. In contrast to the overlying Conemaugh formation, no zones of red clay and shale are present in the Allegheny formation, however. Below the Allegheny formation is the Pottsville formation whose thickness can range from 60 – 440 feet. In the Youghigheny basin, the Pottsville formation is approximately 250 feet thick and is made up of sandstones interbedded with thin silt stones and shales and few coal seams. The basal unit of the formation is characterized by massive sandstones and conglomerates. These resistant sandstones are responsible for the ridges in the area (e.g., Big Savage Mountain, Backbone Mountain, and Winding Ridge).

Murley Run and Bull Glade Run overly the Allegheny/Pottsville formation in their

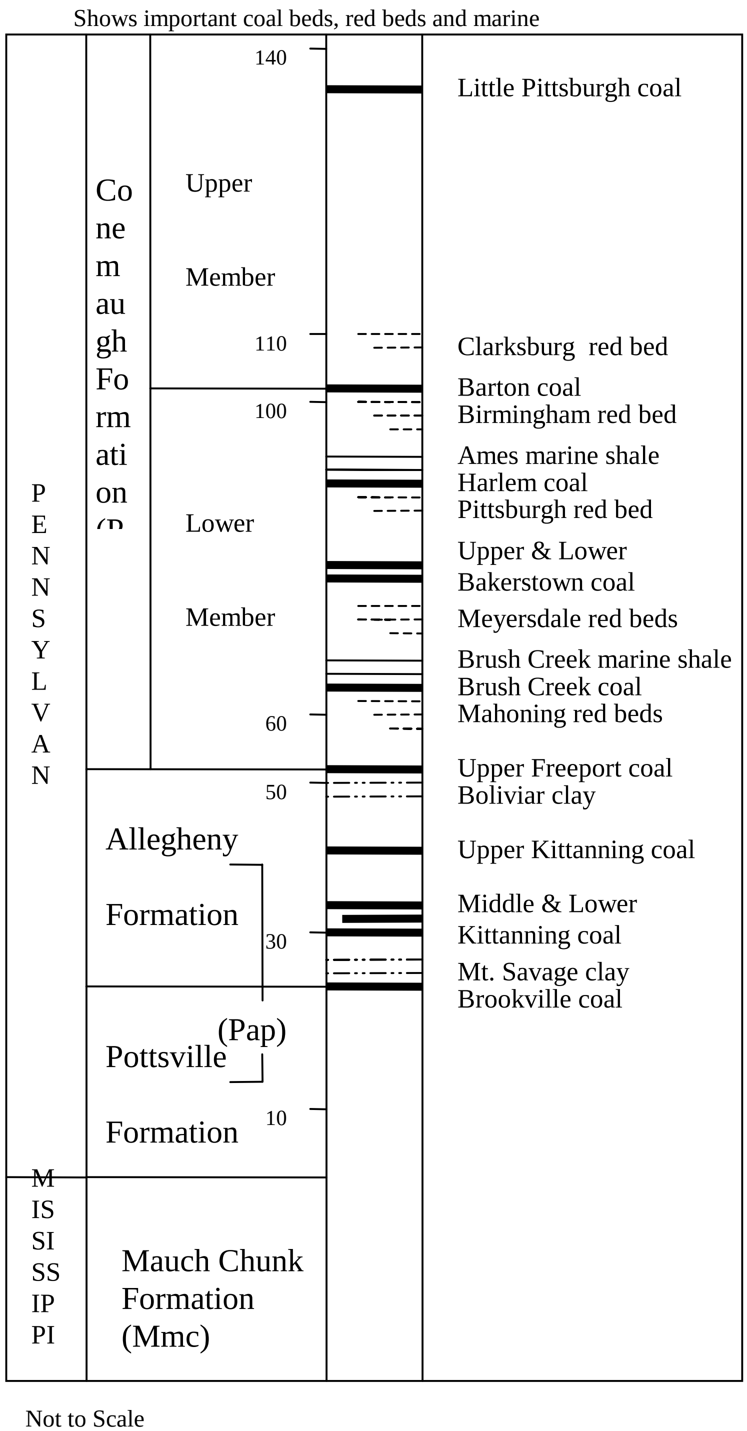

Figure 21. Major geological strata (Pennsylvania period) in the Murley/Bull Glade Run area. Figure adapted from Amsden (1953).

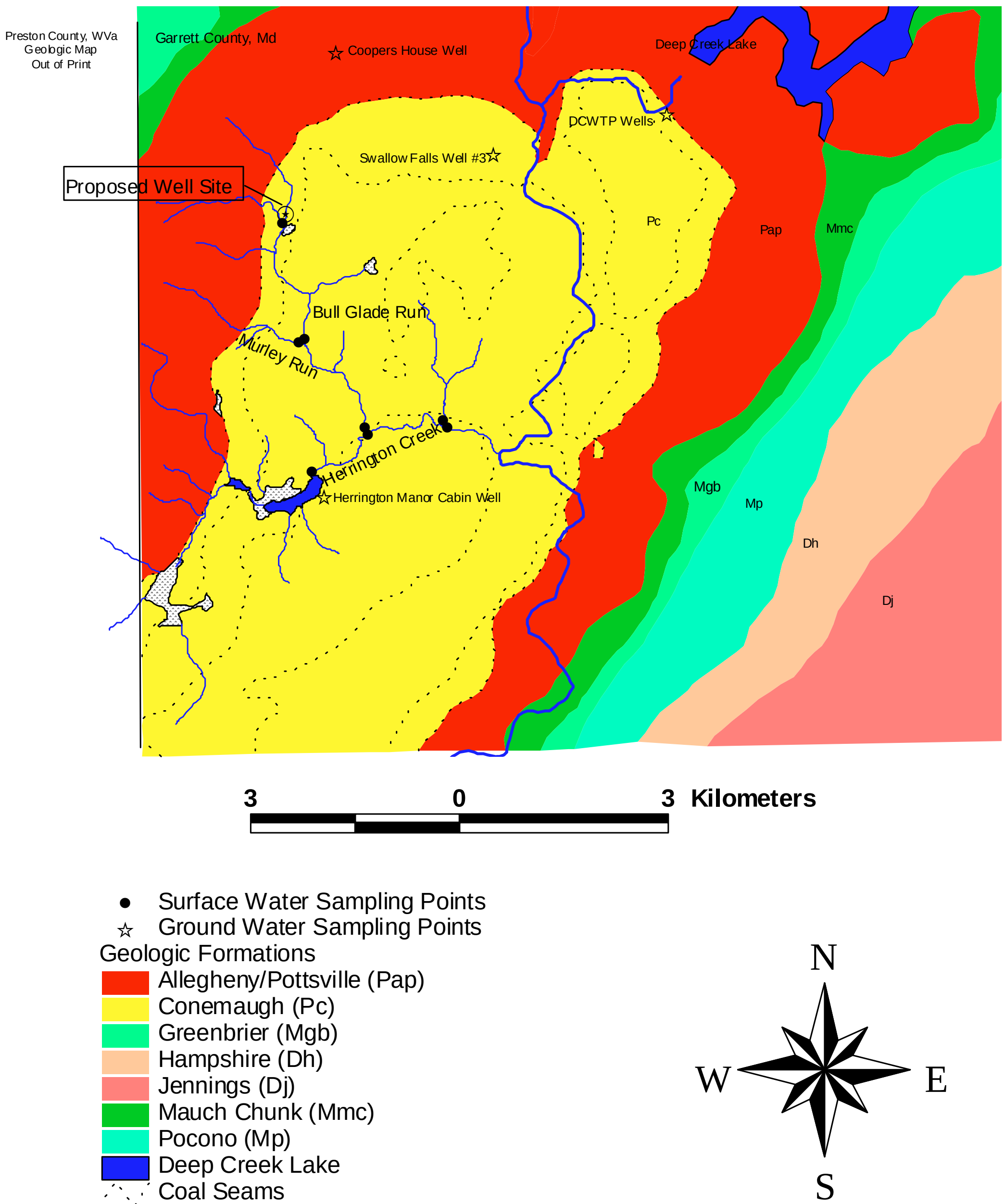

Figure 22. Geological map of the Murley/Bull Glade run area (redrawn from Amsden, 1953).

uppermost reaches and the Conemaugh strata in the lower reaches (Amsden, 1953; Figure 21). The proposed well site, while at the western edge of the Conemaugh formation in the Upper Youghigheny basin, is only 0.3 km from the Conemaugh and Allegheny/Pottsville formations' border. Given the errors associated with geologic mapping at the scale of the map (Garrett County, 1:62,500), it cannot be stated with 100% certainty which is the uppermost geologic formation at this location. However, even if the Conemaugh formation overlies the Allegheny formation at this point, the Conemaugh strata has been significantly eroded in the Upper Youghigheny basin and, in places, may be totally absent (Vocke & Edwards, 1974). For this reason, a deep well at the proposed site will most probably produce the quality and quantity of groundwater associated with the Allegheny formation.

To "ground truth" the geologic map and gain more site specific information, State of Maryland Water Resources Administration well completion reports were obtained from either from the Garrett County Health Department, local well drillers, or Maryland Environmental Services for five wells within a 5 km radius of the proposed well site (Figure 22 and Table 2). According to Amsden (1953), the Cooper house well located to the north is drilled into the Allegheny/Pottsville formation and Deep Creek Wastewater Treatment Plant (DCWTP) cased and uncased wells are located at the border of the Conemaugh and Allegheny/Pottsville formation. Red shales are noted at the DCWTP-uncased well site and red, dark gray and very black shales and a thin coal bed (at 160' depth) are noted at the Cooper house well site. While thin coal beds are associated with both the Allegheny and Conemaugh formations, the dark and red shales are consistent with the upper member of the Conemaugh formation. According to the well completion reports, water was encountered at 200-250 feet below the ground surface at both sites in grey sandstone and shales. The Coopers house well and the DCWTP-cased well were

Table 2. Groundwater Well Details (depths refer to distance, in feet, below the ground surface).

Well Identification	Total Depth	Sealed Depth	Hole Open	Geologic Formation (Vocke & Edwards, 1974)	Date Sampled	Pump Capacity (GPM[1])
Herrington Manor State Park cabin well	357	0-67	68-357	Conemaugh	10/13/99	12
Swallow Falls State Park well #3	310	0-55	56-310	Conemaugh	10/13/99	20
Cooper house well	503	0-166	167-503	Allegany/Pottsville	2/24/00	8
Deep Creek Wastewater Treatment Plant-uncased well at gate	288	0-39	40-288	Allegany/Pottsville	2/24/00	20
Deep Creek Wastewater Treatment Plant-cased well	640	0-229	230-640	Allegany/Pottsville	2/24/00	5

[1]GPM = gallons per minute

designed to yield water with as low an iron concentration as possible (Mike Cooper and Alan Festerman, personal communications). To accomplish this, the wells were sealed to a depth of at least 165 feet, thereby excluding groundwater from the thin coal seams in the shallower strata. The DCWTP-uncased well was not designed in this manner and is sealed only to a depth of 39 feet.

Well completion reports were also secured for two other wells: Swallow Falls well #3 and Herrington Manor cabin well. Both wells were drilled into the Conemaugh formation and encountered water-bearing strata at approximately 125' and 150' below the ground surface, respectively. Water is first encountered in gray “sand rock” and “gray rock”, and then at deeper depths in both wells in dark gray and red shale beds. A thin coal bed was noted at approximately 100 feet below the ground surface at Swallow Falls well #3. Since water from these two wells goes through a treatment process (iron removal, chlorination and pH adjustment) prior to consumption, these wells have very shallow casings of 50-70 feet, leaving 250-300 feet of open hole from which water is pumped.

Vocke & Edwards (1974) report that the shales, sandstones and limestones in western Maryland contain water under artesian and water-table conditions. Although the sandstones are generally massive and relatively impervious, joints and fractures can transmit considerable amounts of water. And although the sandstones are the principal water bearing units, fractures along the shale and coal beds in Garrett County can also produce large amounts of water locally. They report further that the Pottsville, Allegheny and lower member of the Conemaugh formations are the most important water bearing units in the area. In general, water from wells in these formations is usually high in iron concentration and low in hardness.

While no formal tests of well yield were performed at these wells, some indication of water availability can be gained. The well completion reports for the two wells sealed/cased to a depth of at least 160 feet indicate that water is produced from a gray sandstone unit of varying thickness. The log for the Cooper house well indicates that production was less than 1 gallon per minute (GPM) in beds of gray sandstone (189-264 feet deep) and gray sandrock (361-419 feet deep). The DCWTP-cased well log cites water from adjacent beds of gray sandstone (259-265 feet deep) and gray shale (266-340 feet deep), but no indication of production is noted. The water level drawdowns measured at both wells during short term pumping tests were severe (to hole bottom after 1-6 hours of pumping at 1-2 GPM). The capacities of the well pumps installed at both sites are less than 8 GPM. This information suggests that the water bearing sandstone present at both sites is not heavily fractured and therefore does not transmit large quantities of water.

Driller's well logs for the three wells with relatively shallow casings (< 70 feet) were also examined. The two wells drilled into the Conemaugh formation (Swallow Falls well #3 and Herrington Manor cabin well) cite the presence of water at two and three locations, respectively. The Swallow Falls record indicates production of 3 and 8 GPM from a grey sandrock bed (121 to

138 feet deep) and at the interface of a dark gray shale bed (175-215 feet deep) and a grey sandstone (215-265 feet deep), respectively. The water bearing units in the Herrington Manor cabin well are grey rock (100-170 feet and 245-357 feet) and red shale (205-215 feet deep). This "grey rock" could be a limestone layer, as Amsden (1953) reports are found in the upper member of the Conemaugh formation. Capacities of the pumps installed at Swallow Falls well #3 and DCWTP-uncased well are both 20 GPM. The drawdowns measured during short-term pumping tests at the Swallow Falls and Herrington Manor wells (after 12 and 6 hours of pumping at 15 and 18 GPM, drawdowns of 230 and 12 feet measured, respectively) were not as drastic as those cited for the wells in the Cooper house well or the DCWTP-cased well. Lower drawdowns during pumping and the presence of limestone or "grey rock" (a more highly fractured and therefore transmissive geologic layer) in the area suggests greater water availability from the strata at these locations.

Groundwater samples were taken from each of the five wells and were analyzed for the same suite of analytes as the monthly stream water samples. Pertinent analytical results are presented in Table 3. The wells at Herrington Manor and Swallow Falls supply drinking water to the cabins and campgrounds within the parks. Maryland Environmental Services, a state agency that provides waste and environmental management services, operates water treatment facilities at both sites. In general, the treatment process was designed to remove the iron, chlorinate and adjust the pH of the "raw" groundwater prior to its distribution to the campers. Results presented for the Swallow Falls and Herrington Manor State Parks correspond to untreated (i.e., raw) water samples. Results from the well samples demonstrate that sources of alkaline groundwater are present in close proximity to the Murley/Bull Glade Run watersheds. ANC from the well samples ranged from 709 to 2324 μeq/L. Additionally, concentrations of

Table 3. Groundwater Well Chemistry Results.

	ANC (µeq/L)	pH	Mg (ppm)	Ca (ppm)	DOC (ppm)	Mn (ppb)	Fe (ppb)
Swallow Falls Raw	709.2	6.34	4.890	8.430	0.2	0.14	59
Herrington Raw	2324.1	7.82	3.460	19.640	0.2	1.23	BDL
GWWTP @ Gate	2132.2	7.11	2.259	10.105	0.8	241	139
GWWTP-Cased Well	2112.8	7.07	2.286	10.281	0.9	247	15.5
Cooper's Well	2208.1	8.10	0.611	3.067	0.2	49	499

iron and manganese were fairly low at most of the wells. Iron concentrations were below the U.S.E.P.A. EcoTox Threshold of 1000 µg/L at all of the wells. Manganese concentrations were high at only two of the wells, exceeding the ET threshold of 80 ppb.

Personal conversations with a professional local well driller, responsible for the installation of over one thousand household and/or public water supply wells in the area over the past twenty years, informed project personnel that the quality and quantity of water desired dictates the well design at the proposed well location (Wayne Bolden, personal communication). Mr. Bolden simply confirmed what the five well completion reports stated. In the Bull Glade/Murley Run area, a shallower well with a shorter cased/sealed length and longer open hole would typically produce a higher yield, but the quality of the water would likely be influenced by the coal seams found in the shallower strata in the area. Water from such a well would probably have a relatively low pH and high iron content, similar in quality to water drawn from Swallow Falls well #3 and Herrington Manor cabin well. A deeper well, with a greater cased or sealed section (like the DCWTP-cased well or Cooper house well) can exclude the water influenced by the numerous coal seams, but produces less water than the shallower well and the water will usually have a higher pH and lower iron concentration. The cost to install a 500 foot deep well and outfit the well with an appropriate capacity pump would cost between $5,000 and $10,000. The exact cost would depend upon the length of casing and the diameter of the well desired.

In order to design an alkaline groundwater pumping system that would be adequate to neutralize the strong acidity of Bull Glade Run, we evaluated the response of the system to two different types of groundwater injection: (1) a constant rate injection; and (2) a variable rate injection that discharged groundwater from a storage reservoir at a constant proportion of the ambient flow. In the latter case we sized a storage reservoir that would be adequate to store an adequate volume of groundwater that allows neutralization to take place under all flow conditions measured during the study period (1996-2000). For both schemes we assumed a constant groundwater ANC (2000 μeq/L) that is close to the mean value shown in Table 3; in addition, since the ANC of Bull Glade Run does not vary dramatically as a function of stream discharge, we used the simplistic assumption of a constant ambient ANC equal to –10 μeq/L for all cases. A streamwater mixing model (coded in MS-Excel) was used to predict the changes in streamwater ANC associated with each injection scheme.

Results for a constant rate groundwater injection of 1.0 L/sec (~16 gal/min) are shown in Figure 23. The results indicate that the stream acidity is substantially neutralized, but not under all conditions; the model predicts that some "episodes" associated with high flow conditions would occur frequently during the winter and spring months at this injection level, although minimum ANC levels would be higher than they are presently. With this scheme, streamwater ANC would be expected to increase dramatically and approach a level of 2000 μeq/L during low flow periods when streamflow becomes dominated by groundwater.

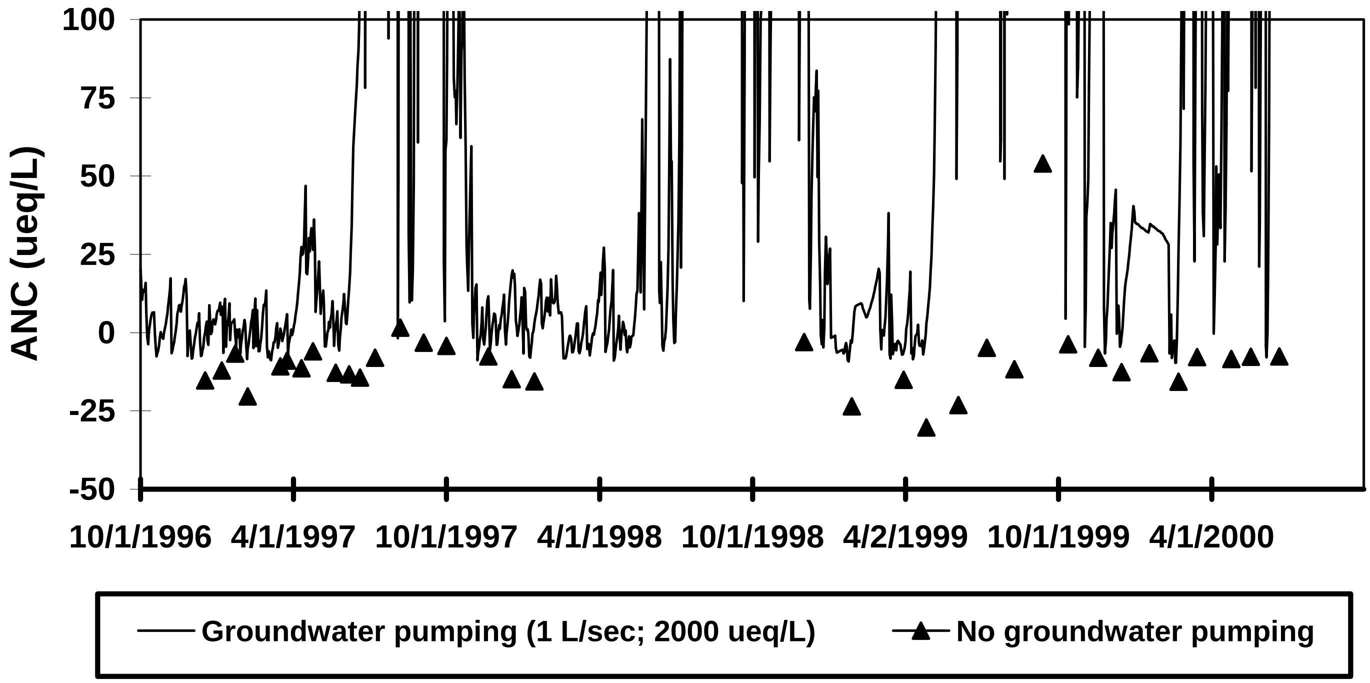

Figure 23. Predicted response in streamwater ANC of Bull Glade Run (Bull) to a constant rate injection of alkaline groundwater (pumping rate = 1.0 L/sec, ANC = 2000 μeq/L) during the period 10/1996 through 6/30/2000. Ambient ANC values without groundwater pumping are shown for comparative purposes.

Results for a variable rate injection scheme using a constant proportion of 0.5% of the ambient streamflow are shown in Figure 24. With this scheme, the spreadsheet model predicts that stream acidity is almost perfectly neutralized under most conditions, although episodes of high ANC occur during conditions in which discharge goes to zero. Recalling that the Bull Glade system is somewhat more tolerant of drought than the HRTB system on which the discharge is based, these high ANC episodes would be expected to be much less important under real field conditions. In order to achieve this level of acid neutralization while extracting groundwater at an average rate of 0.5% of the ambient flow (0.61 L/sec = 10 gal/min), it would obviously be necessary to utilize a reservoir to provide storage for the groundwater influent. We also used the MS-Excel mixing model to size such a storage reservoir and the results of the study are shown in Figure 25. Using an initial storage reservoir of 15000 L (~4000 gal), the injection system is theoretically capable of achieving the results shown in Figure 24; no storage deficits are evident over the 4-year period. (Note: a 4,000 gallon cylindrical tank with a depth of 4 ft has a diameter of about 13 ft.)

We examined the same results when the groundwater injection rate was increased to 1.0% of the ambient streamflow (1.21 L/sec = 19 gal/min). In this example, the model predicts that streamwater ANC could be increased to about 10 μeq/L under virtually all conditions, with the same high ANC episodes still evident in the model predictions (Figure 26). The rate of groundwater pumping in this example is doubled, requiring a storage reservoir that is twice as large (30,000 L = 8000 gal) to accomplish the acid mitigation (Figure 27). Not only is the storage reservoir necessary for continuous acid neutralization under rapidly changing flow conditions, but the storage reservoir (if it could be engineered as a gravity-fed system) could also be used to guard the system against problems caused by electrical power outages which would

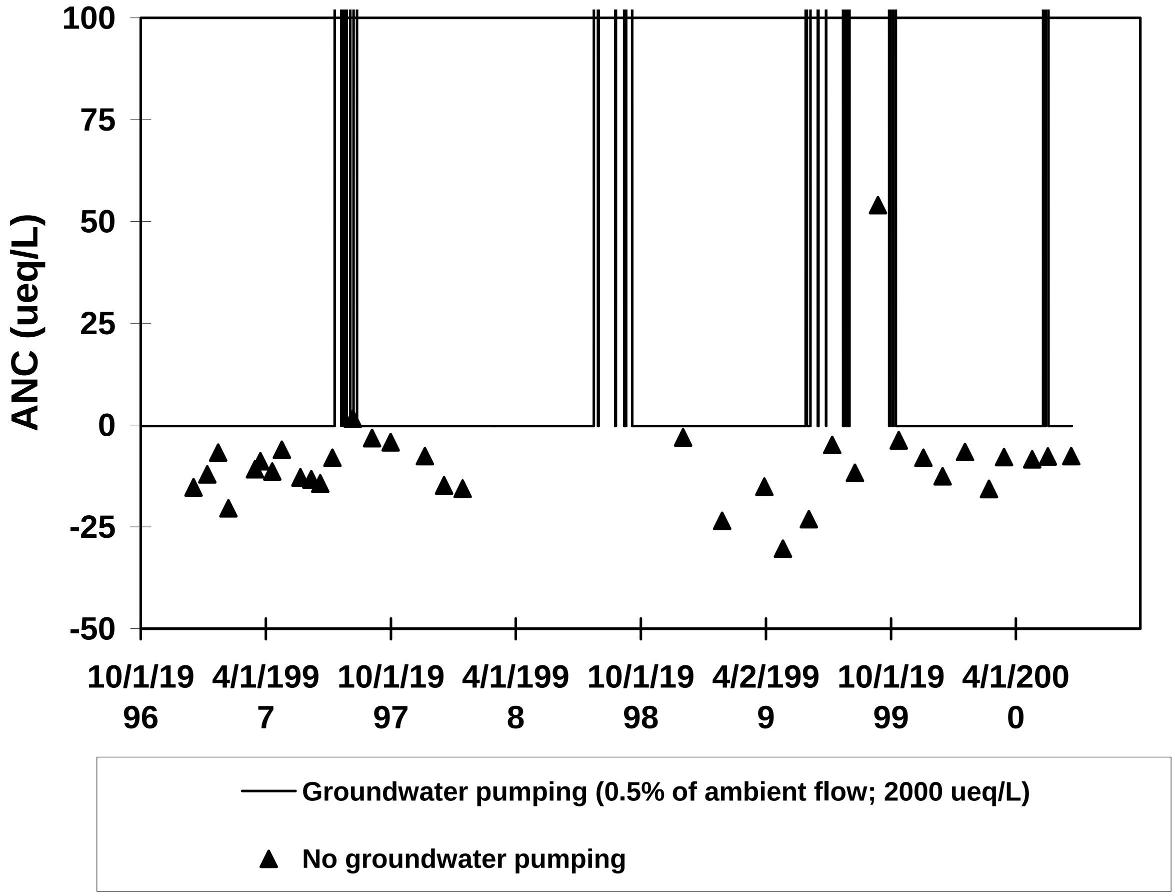

Figure 24. Predicted response in streamwater ANC of Bull Glade Run (Bull) to a variable rate injection of alkaline groundwater (pumping rate = 0.5% of ambient stream discharge, ANC = 2000 μeq/L) during the period 10/1996 through 6/30/2000. Ambient ANC values without groundwater pumping are shown for comparative purposes.

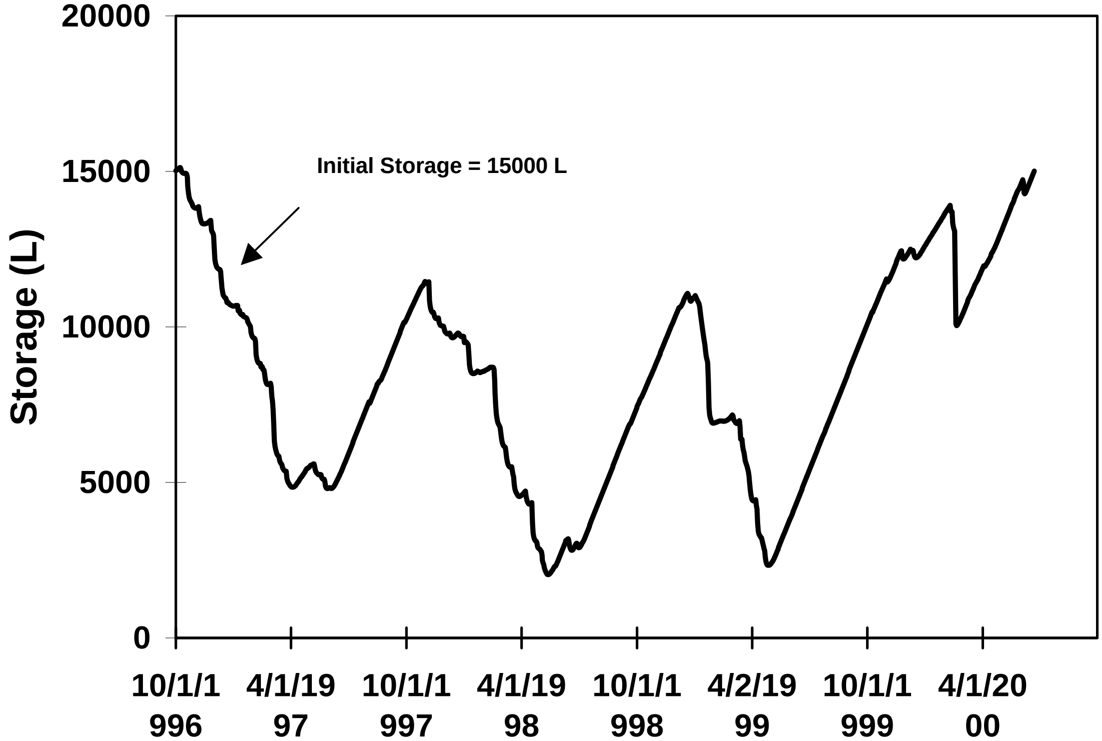

Figure 25. Storage reservoir changes associated with the variable rate injection of alkaline groundwater (pumping rate = 0.5% of ambient stream discharge) to Bull Glade Run during the period 10/1996 through 6/2000.

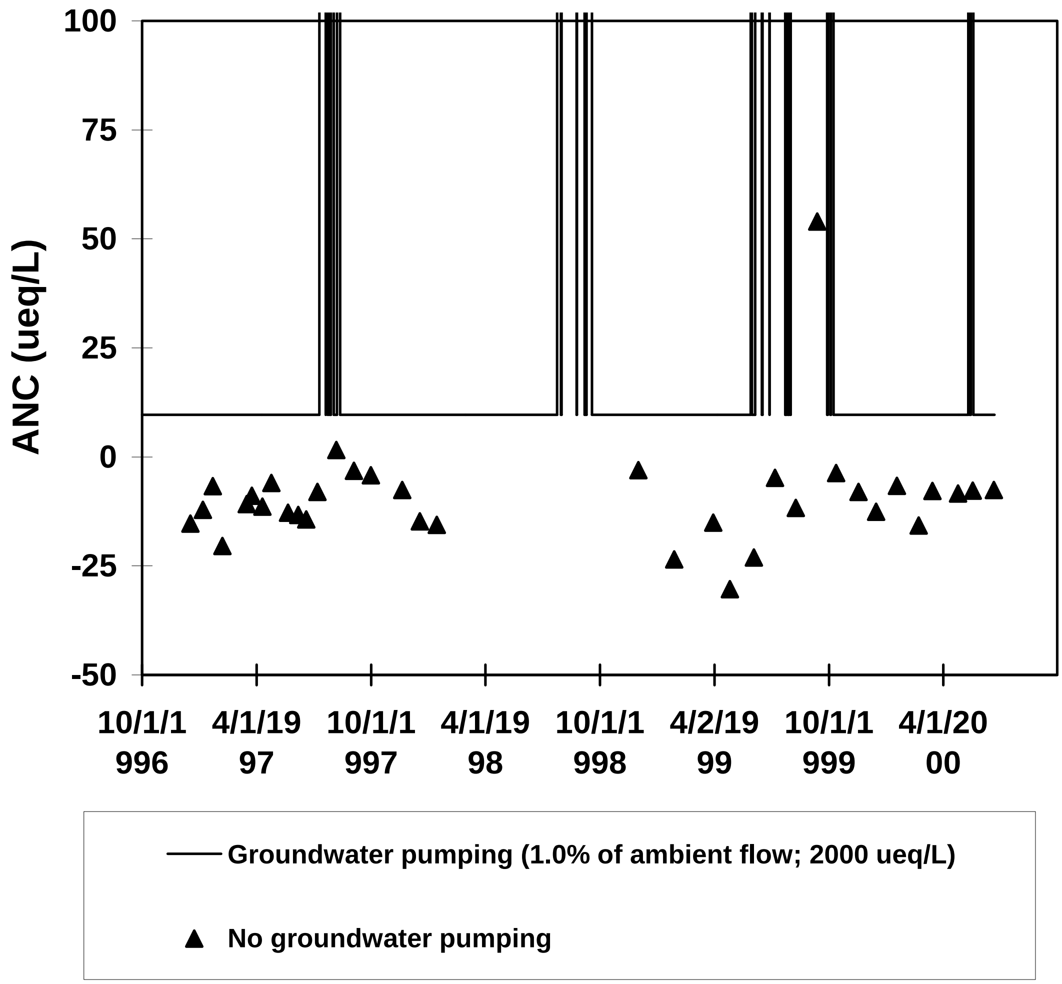

Figure 26. Predicted response in streamwater ANC of Bull Glade Run (Bull) to a variable rate injection of alkaline groundwater (pumping rate = 1.0% of ambient stream discharge; ANC = 2000 μeq/L) during the period 10/1996 through 6/30/2000. Ambient ANC values without groundwater pumping are shown for comparative purposes.

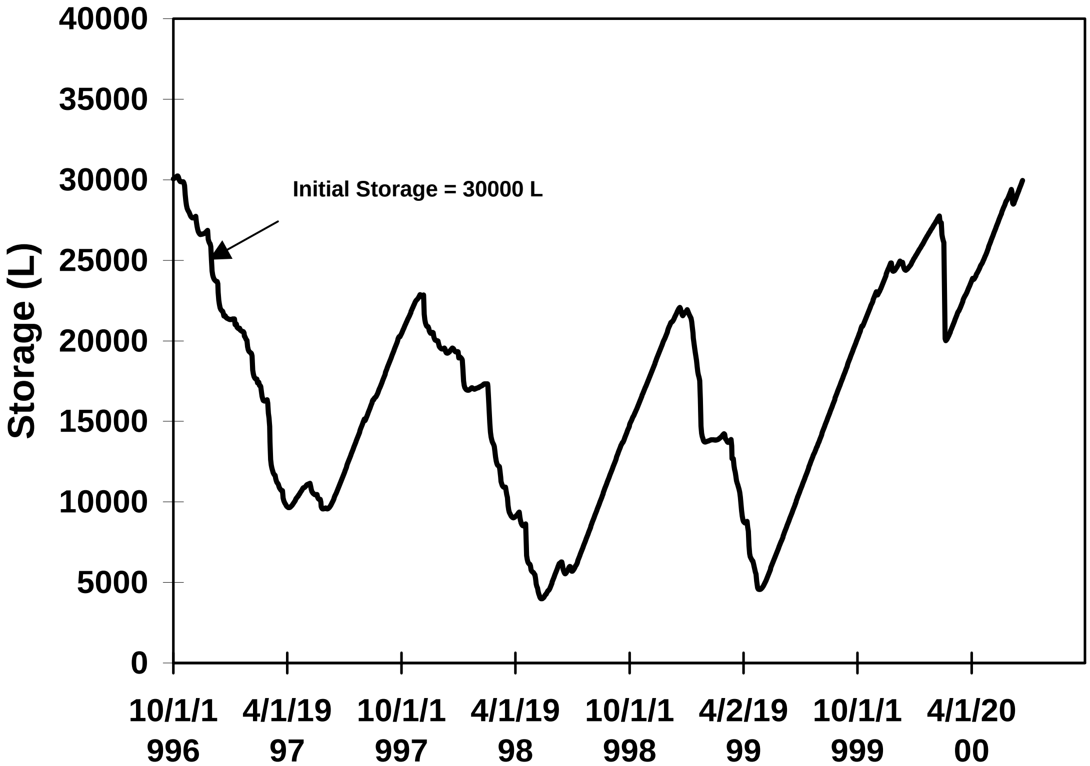

Figure 27. Storage reservoir changes associated with the variable rate injection of alkaline groundwater (pumping rate = 1.0% of ambient stream discharge) to Bull Glade Run during the period 10/1996 through 6/2000.

cause pump shutdowns. (An 8,000 gallon cylindrical tank with a depth of 4 ft has a diameter of 18.5 ft and is about the size of an average backyard swimming pool.)

As a check on the model computations, we also estimated the volume of groundwater that would be needed to exactly neutralize the acidity present in Bull Glade Run. We assumed an average annual runoff rate from the HRTB site of 711 mm/yr, very close to the long-term annual averages for three U.S.G.S. gaging stations in the Monongahela drainage basin (Youghiogheny River at Friendsville, MD: 759 mm/yr; Bear Creek at Friendsville, MD: 655 mm/yr; and Casselman River near Grantsville, MD: 665 mm/yr). Using the mean ANC of –10 μeq/L, the acidity to be neutralized in an average year is 4.81×10^4 equivalents. Assuming a groundwater ANC of 2000 μeq/L, the groundwater volume required to produce a solution with ANC equal to zero is 0.76 L/sec (12 gal/min)—a value that is within the range of the values used in the model runs. The computed result thus supports the output from the dynamic modeling effort.

In-stream Addition of Limestone Sand (ILS)

In-stream addition of limestone sand (ILS) involves the addition of limestone sand directly into the stream channel using dump trucks and/or helicopters. This technique has been effective in West Virginia, Virginia, and Pennsylvania as a treatment for streams impacted by acid deposition and/or acid mine drainage. Various strategies have been suggested for determining appropriate dosage amounts for ILS. Most are based on some combination of watershed size, sulfate loadings, and current water chemistry. The method developed by Downey *et al.* (1994) seems to be the most sensitive and allows for prediction of expected ANC increases. It uses sulfate loadings to estimate pre-industrial age ANC levels, allowing for more precise dose determinations rather than more gross treatments that result in marked ANC increases to levels that may never have been experienced by the treated stream. Applying the calculations to Bull

Glade above its confluence with Murley Run, the amount of limestone that would be required to raise the ANC by 150 µeq/L can be calculated. Annual discharge from the watershed can be calculated using watershed area (676 hectares) and average annual rainfall (1.0 m) for the area. Using the 63% yield calculated for HRTB (Eshleman *et al.*, 2000), in one year a total volume of 4.26×10^8 L is discharged from Bull Glade. Using this volume and a targeted ANC increase of 150 µeq/L, approximately 3.1×10^3 kg (3.4 tons) of calcium carbonate would need to be dissolved annually. Since only sulfate loadings are incorporated into the calculations, allowances must be made for nitrate loadings within the watershed, as well. Additionally, variations in discharge and deposition could be expected. Therefore, approximately 5.0×10^3 (5.5 tons) would need to be added to the stream according to the methods developed by Downey *et al.* (1994).

Researchers at Pennsylvania State University employed less sophisticated technique to calculate doses (LeFevre and Sharpe, 2000). By multiplying the watershed area in acres by a constant of 0.05, about 84 tons of calcium carbonate would need to be added to Bull Glade for successful treatment. In the first year of treatment, dosage is doubled to allow adequate treatment.

Another technique uses stream acidity, discharge and a tonnage factor of 0.086 to estimate limestone dosage requirements (Sampsell, 1999). Using the average acidity measurements from the effluent water collected from Bull Glade for the column experiments (Gannett Fleming, 1997) and the annual average discharge from 1999-2000 (0.15 m^3/sec), approximately 40 tons of calcium carbonate would be the required dose for restoration.

Considering the disparate doses calculated using the different techniques employed in the past, it is difficult to determine actual required doses to achieve adequate improvements in ANC. At the very maximum, a dosage of about 80 tons annually could be expected. Limestone can be purchased at about $9.50 a ton, which would result in a cost of about $1500 for the first year treatment and about $750 each subsequent year to allow treatment of Bull Glade Run.

ILS could potentially be a more cost-effective and logistically feasible alternative to alkaline groundwater pumping. This technique requires very little maintenance and the estimated costs associated with it are much lower. Additionally, the "foot print" for such a restoration effort would be quite minimal. Dosage is typically performed using dump trucks, but helicopters have also been used to restore more remote locations. Therefore, the only requirement would be access to the stream channel.

Discharge and gradient characteristics of the treatment stream are important to consider because stream flow is used to disperse the neutralizing material downstream. Downey *et al.* (1994) found that with lower gradient streams, the sand dispersed further downstream than at other treatment sites, but there were long segments in the stream where sand did not become entrapped. Since Bull Glade is a fairly small stream with an extremely low gradient, it is difficult to predict the efficacy of such a treatment method. Additionally, the presence of a significant number of beaver impoundments and wetlands within the Murley/Bull Glade Run system would be expected to further retard the movement of sand throughout the stream system.

Constructed Wetland

Construction of a treatment wetland has been proposed in the past for the Murley/Bull Glade Runs watershed (Gannett Fleming, 1997). Constructed wetlands have been most commonly used to treat acid mine drainage (AMD). Review of the literature has not yielded evidence of success in using wetlands for the treatment of acidic deposition. In AMD-treatment wetlands reductions in acidity are usually secondary to the main function, which is precipitation of manganese and iron hydroxides (i.e., "yellow-boy”) from mine effluent. Anoxic limestone drains (ALD) are typically used when acidity reductions in AMD are needed.

Construction of a wetland would also involve considerable cost and disruption to the watershed. Installation of a constructed wetland is environmentally disruptive, requiring the use of heavy excavating machinery to properly grade the large quantities of limestone gravel and mushroom compost that must be transported to the site. In the case of the Murley Run watershed, a proposed artificial wetland would also leave a large "footprint" on the landscape (minimum of 0.22 hectare). Most treatment wetlands can only handle relatively small influent discharge rates, so the effects of episodic events on stream water quality would need to be considered in relation to this constructed wetland, especially since substantial depressions in pH and ANC are often seen during periods of high flow. Although column experiments have been performed to estimate the required wetland size to achieve adequate improvements in water quality (Gannett Fleming, 1997), there is still no guarantee that one wetland area would provide sufficient water quality improvements in the short term; in fact, mention is made in the cited report that funds should be reserved by the state for a second wetland area to be constructed to complete the mitigation project. Finally, the efficiency of most treatment wetlands has been shown to decline rapidly with age, thus requiring relatively frequent "renovations" to enhance their water quality mitigation functions.

Conclusions and Recommendations

Our analysis suggests that of the three possible mitigation strategies considered in this report, only alkaline groundwater pumping and instream addition of alkaline sand are technically viable as long-term solutions to the acidification problem in the Murley Run watershed. Based on our review of other studies, we do not believe that a constructed wetland could adequately restore ANC to surface waters in this watershed; in addition, a constructed wetland would require a relatively large area of the forest and would require frequent excavation and renewal—two major disadvantages of this technique.

On the other hand, an alkaline groundwater pumping scheme—combined with an adequately sized storage reservoir—could effectively restore positive ANC to Bull Glade Run. This technique has the major advantage that streamwater ANC could be closely controlled by varying the rates of groundwater input to the stream as described in the earlier sections of the report. However, this technique would be estimated to require about $80,000 in up-front construction costs ($60,000 for electrical power transmission to the site, $10,000 for well-drilling, and $10,000 for a pump, storage tank and ancillary equipment). In addition, we recommend siting a continuously-recording stream gage and water quality monitoring station (equipped with a programmable sequential water sampler) at the Bull site to monitor changes in acid-base chemistry both before and during the period of acid mitigation. We estimate that the gaging/monitoring station could be installed for an additional $10,000 for a total construction cost of $90,000.

In addition, the alkaline groundwater mitigation scheme would require substantial use of electrical power and routine maintenance in perpetuity. We estimate that electrical power costs could run as high as $4,000/year with maintenance costs of about $2,000/year. The scheme would also require water quality sampling at relatively high frequency to demonstrate the efficacy of the mitigation effort—particularly during the first year of the project. We recommend that samples be collected daily during the first year (and also during some high discharge events), but this sampling rate could be reduced to weekly in later years after the system has been adequately calibrated. All samples should be analyzed for pH, ANC, total reactive Al, dissolved Mn, and dissolved Fe using the methods described previously. Biological monitoring of fish and benthic invertebrate communities could occur with lower frequency such as two times per year. The costs of the water quality and biological monitoring would be approximately $15,000 in the first year and $10,000 in subsequent years. The total project cost for mitigating acidity in Bull Glade Run over a 10-year period would be about $265,000.

In contrast, the in-stream alkaline sand treatments could be accomplished at a much lower cost. Assuming that the sand treatments could be completed using dump trucks, the first year cost could be as low as $1,500 with subsequent annual treatments running $750/year. Combined with the water quality and biological monitoring costs, the total project cost over a 10-year period would be about $134,000 or just about half of the cost of the groundwater mitigation project.

This study has provided MDNR with a baseline water quality and biological data base for the Murley/Bull Glade watershed that is an essential component of any mitigation project. In the event that an acid mitigation strategy is actually proposed, funded and implemented, the data are now available for comparison against future monitoring information. Three approaches to acid mitigation were discussed in this report and two are believed to be feasible alternatives should the State of Maryland wish to consider embarking on such a project. We believe that this type of local acid mitigation should be implemented in the form of a demonstration/research project—from which both state environmental regulators and researchers could gain substantial experience with a relatively novel type of water quality restoration. With the help of local contractors in western Maryland, Appalachian Laboratory and MDNR would possess all of the necessary hydrological, water quality monitoring, and biological monitoring expertise to fully design and implement such a project in the event that a decision is made to pursue this type of mitigation activity in the future.

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Appendix A (Tables A1—Tables A6)

Table A1. Numbers of macroinvertebrates in benthic samples collected by combining 9 D-frame aquatic net samplings (total sampling area approximately 1 m²) on 27 April and 28 April, 2000 in unnamed tributaries to Murley Run Insect quantities represent numbers of larvae or nymphs unless designated otherwise by a P for pupa or A for adult. The first number is a total, followed by the number of individuals in a particular life stage, if other than larva or nymph. *specimens were either too small, damaged, or in an inappropriate life stage for further identification.

Taxon **Station**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**9**	**10**	**11**	**12**	**13**	**15**
Turbellaria													
Phygocata sp.	1	1	3	1		6							
Nematoda			1										
Oligochaeta													
Lumbriculidae													
Eclipidrilis sp.	20	1	9	11	9	25	30	10	10	6	11	4	2
Naididae		1										1	
Tubificidae		4	1	3	2			1		1		6	5
Crustaceae													
‘Hydracarina'		1		1					1	3			
Isopoda													
Asellidae													
Caecidotea sp.		1	1			6							
Decapoda													
Cambaridae *			1	1				1			1		
Insecta													
Collembola		2			1								
Ephemeroptera													
Ameletidae													
Ameletus lineatus		4	4	1						3			1
Baetidae													
Baetis sp.												1	
Ephemerellidae													
Ephemerella doris/temporalis													3
Heptageniidae													1
Odonata													
Aeshnidae													1
Aeshna umbrosa													1
Plecoptera													
Leuctridae													
Leuctra sp.	182	156	156	119	147	125	174	278	172	221	176	249	245
Nemouridae *				1		1							4
Amphinemoura wui	66	79	91	144	45	90	16	24	37	67	62	38	4
Osterocera sp.	2	12		5	3	23	3	1	26	19	7	2	10

Table A1 (continued).

Taxon **Station**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**9**	**10**	**11**	**12**	**13**	**15**
Peltoperlidae													
Talloperla sp.		6	3		1	4			3				
Perlodidae													
Cultus sp.												1	
Isoperla cotta								1					
Isoperla sp.						1							
Megaloptera													
Corydalidae													
Nigronia serricornis								2					1
Sialidae													
Sialis aqualis							1						
Trichoptera													
Hydropsychidae													
Diplectrona modesta					1			1				2	8
Hydroptilidae													
Palaegapetus celsus	3			1						1			
Lepidostomatidae													
Lepidostoma sp.	1		3			1				2			
Leptoceridae													1
Philotomatidae													
Wormaldia sp.													1
Rhyacophilidae													
Rhyacophila minor					1		7	5			2	4	
Rhyacophila nigrita	8	5	11	10	8	22	3		6	7	4		
Unenoidae													
Neophylax mitchelli	1		2	1	2	2	2		5				
Lepidoptera													
Pyralidae													
Synclita sp.								1					
Coleoptera													
Elmidae			1										
Diptera													
Ceratopogonidae													
Bezzia sp.		1	7	3	1				3	2		1	
Dasyhelea sp.		1										2	
Palpomyia/Sphaeromias sp.			1										

Table A1 (continued).

Taxon Station	1	2	3	4	5	6	7	9	10	11	12	13	15
Chironomidae									8				
Tanypodinae	5	1	7	4	2	*3*	2		7	3	1	1	5
Orthocladinae	4	5	12	*5*	3	*3*	3		17	6	1	3	10
Chironominae	2	1	4						1	1		1	
Tanytarsisni	1	5	17	*5*	*2*	1	2		4	6	1		6
Empididae													
Chelifera sp.	1	1			1				1				
Clinocera sp.			1										
Hemerodromia sp.								2			1	1	1
Simulidae													
Prosimulium sp.		1	3	34	65	4	53	6		*23*	25	4	1
Stegopternata mutata mutra	19	48		6	19	3	27	1	3	*7*	6		1
Tabanidae													
Chrysops sp.													1
Tipulidae													
Dicranota sp.	1					2			1	2			
Hexatoma sp.	1	1					1					1	
Tipula sp.													1

Table A2. Numbers of benthic macroinvertebrates collected in Course Particulate Organic Matter (CPOM) samples on 27 April and 28 April, 2000 in unnamed tributaries to Murley Run Insect quantities represent numbers of larvae or nymphs. *not included in calculations.

Functional Feeding Group	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**9**	**10**	**11**	**12**	**13**	**15**
Shredders													
Plecoptera	859	226	220	390	226	256	220	228	273	212	137	286	125
Other				1									2
Scrapers	0	0	0	0	0		0				0	0	
Ephemeroptera													2
Other						1			1				1
Filtering Collectors							0						
Trichoptera		1	0		1								
Diptera	68	54		68	54	5		1	1	31	68	2	30
Other													
Gathering Collectors													
Oligochaeta	25	1	4		1	7	4		9			2	1
Diptera	110	3	48	46	3	55	48	2	24	6	9	2	24
Other				2							2		1
Predators													
Plecoptera						1		3					
Trichoptera		3		5	3	10		1	2	8	3		
Diptera	81		24	8		21	24	1	9	2	5	3	3
Other	25					6							
Other Macroinvertebrates	4		1				1		2		1	1	

Table A3. Data summary of benthic macroinvertebrates collected in D-frame aquatic net samples on 27 April and 28 April, 2000 in unnamed tributaries to Murley Run. The metrics for the riffle community are: taxa richness = total number of taxa recognized; total EPT taxa = total number of recognized taxa of Ephemeroptera, Plecoptera, and Trichoptera; Ephemeroptera taxa = number of mayfly taxa; Diptera taxa = number of "true" fly taxa (including midges); % Ephemeroptera = percent mayflies nymphs; % Tanytarsini = percent of Tanytarsini midges to total fauna; intolerant taxa = number of taxa considered to be sensitive to perturbation (Hilsenhoff values 0 - 3 (Hilsenhoff, 1987)); % tolerant = percent of sample considered tolerant of perturbation (Hilsenhoff values 7 - 10 (Hilsenhoff, 1987)); % collector gatherers = percent of sample that feeds on detrital deposits or loose surface films. These metrics were then scored according to the following criteria: taxa richness: >22 = 5, 16-22 = 3, <16 = 1; total EPT taxa: >12 = 5, 5-12 = 3, <5 = 1; Ephemeroptera taxa: > 4 = 5, 2-4 = 3, < 2 = 1; Diptera taxa: > 9 = 5, 6-9 = 3, < 6 = 1; % Ephemeroptera: >20.3 = 5, 5.7-20.3 = 3, < 5.7 = 1; % Tanytarsini: > 4.8 = 5, > 0.0-4.8 = 3, 0.0 = 1; intolerant taxa: > 8 = 5, 3-8 = 3, < 3 = 1; % tolerant: < 11.8 = 5, 11.8-48.0 = 3, > 48.0 = 1; % collector gatherers: > 31 = 5, 13.5-31.0 = 3, < 13.5 = 1. The Index of Biotic Integrity (IBI) score was obtained by totaling and then averaging the score for each station. That value was assigned into the following ranges: 4.0-5.0 = good; 3.0-3.9 = fair; 2.0-2.9 = poor; 1.0-1.9 = very poor.

Station	CPOM COMMUNITY	RIFFLE COMMUNITY									
	% Shredders	Taxa Richness	EPT Taxa	Ephemeroptera Taxa	Diptera Taxa	% Ephemeroptera	% Tanytarsini	Intolerant Taxa	% Tolerant Taxa	% Collector Gatherers	IBI
1	73.42%	17	7	0	8	0.00%	0.31%	6	13.84%	8.49%	2.33
2	73.59%	23	6	1	10	1.18%	1.48%	6	17.46%	7.10%	2.78
3	73.09%	22	7	1	8	1.18%	5.01%	7	6.19%	14.45%	3.00
4	79.80%	18	7	1	6	0.28%	1.42%	6	16.29%	7.30%	2.33
5	78.47%	18	8	0	7	0.00%	0.62%	8	31.30%	5.43%	2.33
6	70.14%	17	8	0	6	0.00%	0.31%	8	12.73%	10.87%	2.33
7	74.07%	14	6	0	6	0.00%	0.66%	6	34.57%	10.80%	2.11
9	96.61%	14	6	0	3	0.00%	0.00%	8	5.39%	3.59%	1.89
10	82.42%	16	6	0	8	0.00%	1.31%	6	6.56%	13.11%	2.56
11	74.56%	18	7	1	8	0.79%	1.58%	6	10.53%	6.05%	2.56
12	60.89%	13	5	0	6	0.00%	0.34%	5	14.43%	4.70%	2.11
13	96.62%	19	7	1	8	0.31%	0.00%	6	4.95%	5.26%	2.33
15	67.20%	21	9	3	8	1.59%	1.91%	10	4.77%	8.92%	3.00

Table A4-1. Numbers of macroinvertebrates in benthic samples collected with a 6-inch "T" sampler (182.4 cm^2) on 27 April and 28 April, 2000 in unnamed tributaries to Murley Run Insect quantities represent numbers of larvae or nymphs unless designated otherwise by a P for pupa or A for adult. The first number is a total, followed by the number of individuals in a particular life stage, if other than larva or nymph. *speciments were either too small, damaged, or in an inappropriate life stage for further identification.

Station	**1**				**2**				**3**				**4**			
Taxon	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**
Turbellaria																
Phygocata sp.															***1***	
Nematoda																
Oligochaeta																
Lumbriculidae																
Eclipidrilis sp.	16	12	4				2	1				1			5	2
Tubificidae		3										1				
Insecta																
Ephemeroptera																
Ameletidae																
Ameletus lineatus										1	1					
Plecoptera																
Leuctridae																
Leuctra sp.	10	9	4	17			21	5	1	12	2	2	3	5	8	14
Nemouridae *																
Amphinemoura wui	7	12	1	5			1	2					10	4	36	28
Osterocera sp.	3	9												2	1	1
Trichoptera																
Hydroptilidae																
Palaegapetus celsus	1	8														
Rhyacophilidae																
Rhyacophila minor																
R.. nigrita				1							1		1		2	
R. sp.	2	1								1						
Unenoidae																
Neophylax mitchelli	4	2	7						2		1		2	1	3	
Diptera																
Ceratopogonidae																
Bezzia sp.	1		1	1											1	

Table A4-1 (continued).

Station	1				2				3				4			
Taxon	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**
Chironomidae										1A						
Tanypodinae							3									1
Orthocladinae		3		1			2					2			1	1
Chironominae							1									1
Tanytarsisni								1		2						
Simulidae						2P										
Prosimulium mixtum grp.	2													1	1	
Stegopternata mutata	4						2								4	
Tipulidae																
Dicranota sp.			1											1		
Hexatoma sp.		2														

Table A4-2. Numbers of macroinvertebrates in benthic samples collected with a 6-inch “T” sampler (182.4 cm^2) on 27 April and 28 April, 2000 in unnamed tributaries to Murley Run Insect quantities represent numbers of larvae or nymphs unless designated otherwise by a P for pupa or A for adult. The first number is a total, followed by the number of individuals in a particular life stage, if other than larva or nymph. *specimens were either too small, damaged, or in an inappropriate life stage for further identification.

Station	5				6				7				9			
Taxon	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**
Turbellaria																
Phygocata sp.					2			2								
Nematoda		23														
Oligochaeta																
Lumbriculidae																
Eclipidrilis sp.	3	357	8	3			10	18	3	1	4	1				2
Tubificidae	2						1			1						
Crustaceae																
‘Hydracarina'		1														
Decapoda																
Cambaridae *																1
Insecta																
Collembola									1							
Ephemeroptera																
Ameletidae																
Ameletus lineatus				1				1		3						
Baetidae											1					
Ephemerellidae																
Eurylophella sp.															1	
Plecoptera																
Leuctridae																
Leuctra sp.	61	32	4	3	36	8	9	41	15	7	82	2	3		18	29
Nemouridae																
Amphinemoura wui				1	3	1	1	3	4							
Osterocera sp.	1	1			5				1			1				
Peltoperlidae																
Talloperla sp.							1									
Megaloptera																
Sialidae																
Sialis sp.															1	

Table A4-2 (continued).

Station	5				6				7				9			
Taxon	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**
Trichoptera																
Hydroptilidae																
Palaegapetus celsus							1	8								
Lepidostomatidae																
Lepidostomata sp.											1					
Rhyacophilidae																
Rhyacophila minor											3				1	
R.. nigrita	1					1	1		1		1					
R. sp.					5			5	1			1				
Unenoidae																
Neophylax mitchelli				1	4	5	1	6	1							
Elmidae																
Oulimnius sp.						1	1A									
Diptera																
Ceratopogonidae																
Bezzia sp.		1						1			1					
Chironomidae																
Tanypodinae		1			1				1		4					
Orthocladinae	4	24							1		5		2	1	12	2
Chironominae	3										2					
Tanytarsisni	2								1		3				2	5
Simulidae																
Prosimulium mixtum grp.	2	1		1					1							
Stegopternata mutata	1								3		2					
Tipulidae																
Dicranota sp.								2								
Hexatoma sp.													1			
Molophilus sp.											1					

Table A4-3. Numbers of macroinvertebrates in benthic samples collected with a 6-inch "T" sampler (182.4 cm^2) on 27 April and 28 April, 2000 in unnamed tributaries to Murley Run Insect quantities represent numbers of larvae or nymphs unless designated otherwise by a P for pupa or A for adult. The first number is a total, followed by the number of individuals in a particular life stage, if other than larva or nymph. *specimens were either too small, damaged, or in an inappropriate life stage for further identification.

Station	**10**				**11**				**12**				**13**			
Taxon	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**
Nematoda													1			
Oligochaeta																
Lumbriculidae																
Eclipidrilis sp.	7	5	3	12	14	5	1	2	6	1		10		1		
Tubificidae		1		1	1										1	
Crustaceae																
'Hydracarina'					2											
Decapoda																
Cambaridae *													1			
Insecta																
Collembola			1													
Ephemeroptera																
Ephemerellidae																
Eurylophella sp.													1			1
Plecoptera																
Leuctridae																
Leuctra sp.	2	7	20	56	34				6	7	6	15	5	4	20	2
Nemouridae																
Amphinemoura delosa						1	4	62								
A. wui	1		7		9			17				1				
A. sp.					5										3	
Osterocera sp.		3	9	1				3	1							
Megaloptera																
Corydalidae																
Nigronia serricornis																1
Sialidae																
Sialis sp.			1	1												
Trichoptera																
Rhyacophilidae																
Rhyacophila minor								2								
R.. nigrita					3	1										
R. sp.								1								
Unenoidae																
Neophylax mitchelli					1	2		2		2						

Table A4-3 (continued).

Station	10				11				12				13			
Taxon	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**	**T-1**	**T-2**	**T-3**	**T-4**
Diptera																
Ceratopogonidae																
Bezzia sp.													1			
Chironomidae													1			
Tanypodinae		2	1	4	2			10				1			1	1
Orthocladinae		1	1	3	2	1		4			1		61	53	71	15
Chironominae					1			5					3	1		
Tanytarsisni				1	3			2				1	7	2	3	2
Simulidae																
Prosimulium mixtum grp.					4											
Stegopternata mutata			3		4			3	1							
Tabanidae																
Chrysops sp.															1	
Tipulidae																
Dicranota sp.	1	2	1		2	1		1								
Hexatoma sp.			2	1											1	
Pseudolimnophila sp.						1										

Table A4-4. Numbers of macroinvertebrates in benthic samples collected with a 6-inch "T" sampler (182.4 cm^2) on 27 April and 28 April, 2000 in unnamed tributaries to Murley Run Insect quantities represent numbers of larvae or nymphs unless designated otherwise by a P for pupa or A for adult. The first number is a total, followed by the number of individuals in a particular life stage, if other than larva or nymph. *specimens were either too small, damaged, or in an inappropriate life stage for further identification.

Station	**15**			
Taxon	**T-1**	**T-2**	**T-3**	**T-4**
Annelida				
Oligochaeta				
Lumbriculidae				
Eclipidrilis sp.			10	3
Tubificidae		1		
Insecta				
Plecoptera				
Leuctridae				
Leuctra sp.	7			1
Diptera				
Ceratopogonidae				
Bezzia sp.	1			
Chironomidae				
Orthocladinae				4
Chironominae	1			
Empididae				
Hemerodromia sp.	1			
Tipulidae				
Hexatoma sp.	1	1		

Table A5. Hill's diversity measures, evenness, species richness, and the proportional numerical dominance of the number of the most abundant organism of each benthic sample collected from stations on Murley Branch on 27 and 28 May 2000. Means and standard errors of these indices for each set of "T" samples are also listed T = 4 inch T-sample. The metrics are: s = total number of taxa recognized; s' = total number of aquatic taxa recognized; N = total number of individual organisms recognized; N' = total number of identified aquatic organisms; N1 = $e^{H'}$; N2 = 1/λ; E_5 = ((1/λ)-1)/($e^{H'}$-1) = N2-1/N1-1; R_1 = Species richness: s'-1/$\log_e$N'; d = Proportional numerical dominance of most abundant taxon; N_i/N'; Mean = average of metric for "T" samples only; SE = standard error; *undefined because of small sample size.

Station	1					2					3				
metric	T-1	T-2	T-3	T-4	Mean ±SE	T-1	T-2	T-3	T-4	Mean ±SE	T-1	T-2	T-3	T-4	Mean± SE
s	10	10	6	5	8±1.52	0	0	7	4	3±1.97	2	5	4	4	4±0.73
s'	10	10	6	5	8±1.52	0	0	7	4	3±1.97	2	4	4	4	4±0.58
N	50	61	18	25	39±11.76	0	0	32	9	10±8.73	3	17	5	6	8±3.64
N'	50	61	18	25	39±11.76	0	0	32	9	10±8.73	3	16	5	6	8±3.35
N1	7.02	7.84	4.56	2.64	5.515±1.37	*	*	3.44	3.16	3.30±0.08	1.89	2.28	3.79	3.78	2.94±0.57
N2	6.03	7.62	4.64	2.05	4.24±1.17	*	*	2.30	3.27	2.79±0.28	3.00	1.79	10.00	7.50	5.57±2.22
E_5	0.84	0.97	1.02	0.64	0.83±0.11	*	*	0.53	1.05	0.79±0.15	2.25	0.62	3.23	2.34	2.11±0.63
R1	2.30	2.19	1.73	1.24	1.76±0.31	*	*	1.73	1.37	1.55±0.10	0.91	1.08	1.86	1.67	1.38±0.26
d(%)	32.00	19.67	38.89	68.00	39.64±11.86	*	*	65.63	55.56	60.60±2.91	66.67	75.00	40.00	33.33	53.75±11.68

Table A5 (continued).

Station	4					5					6				
metric	**T-1**	**T-2**	**T-3**	**T-4**	**Mean ±SE**	**T-1**	**T-2**	**T-3**	**T-4**	**Mean ±SE**	**T-1**	**T-2**	**T-3**	**T-4**	**Mean±SE**
s	4	6	11	7	7±1.70	10	9	2	6	7±2.08	7	5	9	11	8±1.00
s'	4	6	11	7	7±1.70	10	9	2	6	7±2.08	7	5	9	10	8±1.00
N	16	14	63	48	35±13.98	80	441	12	10	136±119.12	56	16	26	88	47±16.00
N'	16	14	63	48	35±13.98	80	441	12	10	136±119.12	56	16	26	87	46±16.00
N1	2.83	4.80	4.67	3.09	3.85±0.60	2.84	2.10	1.89	5.17	3.00±0.87	3.50	3.42	5.01	5.16	4.27±0.47
N2	2.45	5.35	2.88	2.40	3.27±0.81	1.71	1.50	1.94	7.50	3.16±1.67	2.33	3.16	4.01	3.63	3.28±0.36
E_5	0.79	1.14	0.51	0.67	0.78±0.15	0.39	0.46	1.06	1.56	0.87±0.32	0.53	0.89	0.75	0.63	0.70±0.08
R1	1.08	1.89	2.41	1.55	1.73±0.32	2.05	1.31	0.40	2.17	1.48±0.47	1.49	1.44	2.46	2.02	1.85±0.24
d(%)	62.50	35.71	57.14	58.33	53.42±6.95	76.25	80..95	66.67	30.00	63.23±13.21	64.29	50.00	38.46	47.13	49.97±5.37

Table A5 (continued).

Station	7					9					10				
metric	T-1	T-2	T-3	T-4	Mean ±SE	T-1	T-2	T-3	T-4	Mean ±SE	T-1	T-2	T-3	T-4	Mean±SE
s	11	4	13	4	8±2.71	3	2	6	6	4±1.19	4	7	11	9	8±1.73
s'	11	4	13	4	8±2.71	3	1	6	5	4±1.28	4	7	11	9	8±1.73
N	34	12	110	5	40±27.81	6	2	35	40	21±11.28	11	21	49	80	40±17.92
N'	34	12	110	5	40±27.81	6	1	35	39	20±11.28	11	21	49	80	40±17.92
N1	7.20	2.93	3.18	3.79	4.28±1.15	2.75	1.00	3.25	2.42	2.36±0.56	2.81	5.61	6.20	2.95	4.39±1.02
N2	4.79	2.75	1.79	10.00	4.83±2.12	3.75	*	2.70	1.77	2.74±0.57	2.50	5.83	4.63	1.96	3.73±1.05
E_5	0.61	0.91	0.36	3.23	1.28±0.76	1.57	*	0.76	0.55	0.96±0.31	0.83	1.05	0.70	0.49	0.77±0.14
R1	3.40	1.21	2.55	1.86	2.26±0.54	1.12	*	1.41	1.09	1.21±0.10	1.25	1.97	2.57	1.83	1.91±0.31
d(%)	44.12	58.33	74.55	40.00	54.25±9.04	50.00	*	51.43	74.36	58.60±7.90	63.64	33.33	40.82	70.00	51.95±10.19

Table A5 (continued).

Station	11					12					13				
metric	**T-1**	**T-2**	**T-3**	**T-4**	**Mean ±SE**	**T-1**	**T-2**	**T-3**	**T-4**	**Mean ±SE**	**T-1**	**T-2**	**T-3**	**T-4**	**Mean±SE**
s	16	7	2	13	10±3.61	4	3	2	5	4±0.75	9	5	8	6	7±0.98
s'	15	7	2	13	9±3.42	4	3	2	5	4±0.75	9	5	8	6	7±22.33
N	88	12	5	114	55±31.52	14	10	7	28	15±5.37	81	61	101	22	66±18.04
N'	87	12	5	114	55±31.40	14	10	7	28	15±5.37	81	61	101	22	66±23.46
N1	7.96	5.47	1.65	5.16	5.06±1.50	3.01	2.23	1.51	2.88	2.41±0.40	2.69	1.73	2.61	3.06	2.52±0.34
N2	5.20	6.00	1.67	3.07	3.99±1.14	3.03	2.05	1.40	2.52	2.25±0.40	1.74	1.32	1.88	2.16	1.78±0.42
E_5	0.60	1.12	1.03	0.50	0.81±0.18	1.01	0.85	0.79	0.81	0.87±0.06	0.44	0.44	0.55	0.56	0.50±0.34
R1	3.13	2.41	0.62	2.53	2.17±0.63	1.14	0.87	0.51	1.20	0.93±0.18	1.82	0.97	1.52	1.62	1.48±22.99
d(%)	39.08	41.67	80.00	54.39	53.79±10.82	42.86	70.00	85.71	53.57	63.04±10.86	75.31	86.89	70.30	68.18	75.17±4.84

Table A5 (continued).

Station	15				
metric	**T-1**	**T-2**	**T-3**	**T-4**	**Mean ±SE**
s	5	2	1	3	3±1.00
s'	5	2	1	3	3±1.00
N	11	2	10	8	8±2.33
N'	11	2	10	8	8±2.33
N1	3.19	2.00	1.00	2.65	2.21±0.54
N2	2.62	*	1.00	3.11	2.24±0.64
E_5	0.74	*	*	1.28	0.67±0.37
R1	1.67	1.44	0.00	0.96	1.02+0.43
d(%)	63.64	50.00	100.00	50.00	65.91±13.65

Table A6. Data summary of benthic macroinvertebrates collected in kick net samples between August, 1991 and September, 1993 in an unnamed tributay to Herrington Creek (Price and Morgan, 1993). The metrics for the riffle community are: taxa richness = total number of taxa recognized; total EPT taxa = total number of recognized taxa of Ephemeroptera, Plecoptera, and Trichoptera; Ephemeroptera taxa = number of mayfly taxa; Diptera taxa = number of "true" fly taxa (including midges); % Ephemeroptera = percent mayflies nymphs; % Tanytarsini = percent of Tanytarsini midges to total fauna; intolerant taxa = number of taxa considered to be sensitive to perturbation (Hilsenhoff values 0 - 3 (*Hilsenhoff, 1987*)); % tolerant = percent of sample considered tolerant of perturbation (Hilsenhoff values 7 - 10 (*Hilsenhoff, 1987*)); % collector gatherers = percent of sample that feeds on detrital deposits or loose surface films. These metrics were then scored according to the following criteria: taxa richness: >22 = 5, 16-22 = 3, <16 = 1; total EPT taxa: >12 = 5, 5-12 = 3, <5 = 1; Ephemeroptera taxa: > 4 = 5, 2-4 = 3, < 2 = 1; Diptera taxa: > 9 = 5, 6-9 = 3, < 6 = 1; % Ephemeroptera: >20.3 = 5, 5.7-20.3 = 3, < 5.7 = 1; % Tanytarsini: > 4.8 = 5, > 0.0-4.8 = 3, 0.0 = 1; intolerant taxa: > 8 = 5, 3-8 = 3, < 3 = 1; % tolerant: < 11.8 = 5, 11.8-48.0 = 3, > 48.0 = 1; % collector gatherers: > 31 = 5, 13.5-31.0 = 3, < 13.5 = 1. The Index of Biotic Integrity (IBI) score was obtained by totaling and then averaging the score for each station. That value was assigned into the following ranges: 4.0-5.0 = good; 3.0-3.9 = fair; 2.0-2.9 = poor; 1.0-1.9 = very poor.

Date	CPOM COMMUNITY	RIFFLE COMMUNITY									
	% Shredders	Taxa Richness	EPT Taxa	Ephemeroptera Taxa	Diptera Taxa	% Ephemeroptera	% Tanytarsini	Intolerant Taxa	% Tolerant Taxa	% Collector Gatherers	IBI
Spring sampling											
Mar 1992	37.50%	20	8	1	6	1.75	0.88	10	2.63	17.54	3.00
May 1992	57.47%	21	10	1	8	0.69	0.69	9	7.64	9.72	2.78
May 1993	36.84%	18	8	1	6	0.76	4.55	8	7.58	29.55	2.78
Fall Sampling											
Aug 1991	5.88%	10	3	0	2	0.00	5.56	2	5.56	16.67	2.11
Oct 1991	25.00%	26	13	4	6	6.36	0.91	11	14.55	19.09	3.67
Sep 1992	13.20%	36	13	3	10	1.46	1.78	10	32.20	51.94	3.89
Sep 1993	9.70%	27	13	7	6	12.35	0.00	11	13.53	22.94	3.67